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Phosphorus Dynamics in Alluvial Landscapes: Implications for Achieving Sustainable Agriculture

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Abstract

Competing interests shared between agricultural sectors, environmental authorities, and the public community can be reconciled through maintaining sustainable agricultural practices. As the Australian dairy industry continues to intensify, the need to improve farm sustainability is mounting, as higher inputs of phosphorus (P) raise a number of concerns pertaining to environmental, economical, as well as ethical agendas. To address this issue, improved nutrient management strategies must be implemented to minimise losses of farm P to the environment.

Alluvial landscapes are often utilised for agricultural production, but, they are frequently mismanaged due to the inherently variable nature of alluvial soils. Thus, this study was designed to investigate the influence of variable alluvial soils on P dynamics, and assess what implications this might have for improving P management strategies. A case study approach was adopted for this study, focusing on one large scale dairy operation, located on the Manning River, on the mid-north coast of New South Wales (NSW). The study investigated variations in soil properties between four major alluvial units present within the study site, including, a modern floodplain (MF) unit and three abandoned terrace units (T1- T3) with increasing height (and inferred tentative age) above the existing channel. This was carried out by extracting a number of soil cores and collecting subsamples at various depths, which were then analysed for a range of parameters.

Soil properties revealed to have the greatest influence on P dynamics were those most strongly correlated with the phosphorus buffering index (PBI); namely, clay content (r_s =0.680; p = < 0.001), sand content (r_s = -0.831; p = < 0.001), mean grain size (r_s = -0.698; p = < 0.001), and oxalate extractable aluminium content (Alox) (r_s = 0.822; p = < 0.001). Variations in these parameters between major alluvial units were discovered to be influencing the cumulative PBI (including all depth ranges sampled) in the following pattern: MF < T1 < T2 ≤ T3. This suggested that paddocks containing two or more alluvial units of variable height are highly susceptible to (P) fertiliser use inefficiencies, which may potentially be resulting in increased P losses. The main finding of this study, therefore, is that P fertiliser management strategies should take into account soil variations between major alluvial units of differential height, as this may have implications for improving fertiliser use efficiencies, in turn, minimising losses of P to the environment.

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PHOSPHORUS DYNAMICS IN ALLUVIAL LANDSCAPES: IMPLICATIONS FOR ACHIEVING SUSTAINABLE AGRICULTURE

by

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A thesis submitted in part fulfilment of the requirements of the Honours degree of Bachelor of Science Advanced (Environment)

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CERTIFICATION

The information in this thesis is entirely the result of investigations conducted by the author, unless otherwise acknowledged, and has not been submitted in part, or otherwise, for any other degree or qualification.

Nicholas Robert van Lijf

26/10/2015

ABSTRACT

Competing interests shared between agricultural sectors, environmental authorities, and the public community can be reconciled through maintaining sustainable agricultural practices. As the Australian dairy industry continues to intensify, the need to improve farm sustainability is mounting, as higher inputs of phosphorus (P) raise a number of concerns pertaining to environmental, economical, as well as ethical agendas. To address this issue, improved nutrient management strategies must be implemented to minimise losses of farm P to the environment.

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To my Pop (1928-2014)

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LIST OF ABBREVIATIONS

ABARES – Australian Bureau of Agricultural and Resource Economics and Sciences

APHA – American Public Health Association

ASRIS – Australian Soil Resource Information System

CV – Critical Value

EAL – Environmental Analysis Laboratory (Southern Cross University)

EC – Electrical Conductivity

Fe_{ox}/Al_{ox} – Oxalate extractable iron (Fe) and aluminium (Al)

FIA – Flow Injection Analysis

GIS – Geographic Information Science/Systems

GPS – Global Positioning System

ICP-AES – Inductively Coupled Plasma Atomic Emission Spectrometry

ICP-MS – Inductively Coupled Plasma Mass Spectrometry

MF – Modern floodplain unit

NSW DPI – New South Wales Department of Primary Industries

NSW OEH – New South Wales Office of Environment and Heritage

OC – Organic Carbon

PA – Precision Agriculture

PBI – Phosphorus Buffering Index

RS – Remote Sensing

T – Abandoned terrace unit

UOW SEES – University of Wollongong School of Earth and Environmental Sciences

WEP – Water Extractable Phosphorus

XRD – X-ray Diffraction

CHAPTER 1

Introduction

1.1. Background

Alluvial landscapes bordering natural freshwater systems are commonly utilised for agriculture, primarily due to the high fertility of alluvial soils as well as water accessibility (Rinklebe *et al.* 2007). Losses of nutrients such as nitrogen (N) and phosphorus (P) via leaching, runoff or erosion from agricultural operations may contaminate receiving water bodies, ultimately leading to eutrophication (Gourley *et al.* 2007b, Burkitt 2014). Farms set in alluvial landscapes, therefore, often pose a threat to the health of nearby fresh water resources.

Dairy farms have frequently been identified as a source of high nutrient (i.e., N and P) exports to aquatic environments (Adams *et al.* 2014). In Australia, the dairy industry continues to intensify, in turn, enhancing the potential for environmental degradation (Gourley & Ridley 2005, Withers & Jarvie 2008, Burkitt 2014). Countries outside of Australia (e.g., throughout Western Europe and Northern America) have previously demonstrated the environmental consequences of prolonged intensified agriculture (Rio *et al.* 2011, Smith *et al.* 2013, Barataud *et al.* 2014). Thus, since the onset of intensification, the Australian dairy industry has received a great deal of public and political attention (Gourley *et al.* 2010), and there have been growing concerns regarding the sustainability of dairy farms as they continue to transform into larger, more productive operations (Gourley & Ridley 2005). Consequently, there has been an ongoing pressure for intensified dairies to be investigated for the potential risk they pose to the environment, and how overall nutrient losses may be minimised (Gourley 2004, Gourley *et al.* 2010, Smith *et al.* 2013, Burkitt 2014).

In order to mitigate the risk of (intensified) dairy operations contaminating nearby fresh water resources, farms must maintain sustainable practices. However, achieving sustainable intensification remains a challenging prospect (Smith *et al.* 2013, Burkitt 2014). Sustainable agriculture may be simply conceptualised as a balance between production levels and

environmental impacts (Hendrickson *et al.* 2008, Smith *et al.* 2013). Maintaining sustainable practices may, therefore, be recognised as having both (long-term) environmental and financial benefits. One example of how farm sustainability may be improved is through optimising nutrient management strategies, most notably, fertiliser use efficiency (McDowell 2012, Simpson *et al.* 2014).

Research efforts aimed at identifying ways to improve fertiliser use efficiency have largely focused on applications of P; this is due to several important reasons (Weaver & Wong 2011, Simpson *et al.* 2014):

(1) Phosphate rock (used to manufacture chemical P fertilisers) is a non-renewable resource and as a result, high quality reserves are becoming increasingly scarce;
(2) Ensuring that future P reserves are adequate for global food securities is critical;
(3) The continually rising cost of P fertiliser; and,
(4) Environmental consequences of over-fertilisation of P resulting in increased losses to receiving water bodies.

Achieving optimised P fertiliser use efficiency requires both an understanding of how 'P dynamics' may vary depending on soil characteristics as well as being able to predict where significant variations in soil characteristics are likely to occur (Iqbal *et al.* 2005). Soil P dynamics may be conceptualised as the movement of P (in the soil) in response to a particular driving force (Fardeau 1995). For this study, soil-P interactions represent the particular driving force of interest (as opposed to gravity, for example). Previous research has revealed that P dynamics may be influenced by a number of common soil characteristics, such as: aluminium (Al) and iron (Fe) oxyhydroxides, clay content, pH, grain size and organic matter (Dougherty *et al.* 2010). Soil characteristics such as these may vary considerably with both depth and location within alluvial landscapes utilised for agriculture (Walker 1989, Iqbal *et al.* 2005). Thus, P dynamics within alluvial landscapes may be harder to predict, in turn, creating an issue for farm managers wishing to optimise their nutrient (i.e., P) management practices. Hence, additional research may be necessary to help determine a range of suitable approaches able to effectively address this issue.

Current available methods proven to improve nutrient management are not widely adopted given the prohibitive costs of the associated infrastructure, machinery, soil mapping data and technology required (Simpson *et al.* 2014, Hedley 2015, Schoumans *et al.* 2015); for

example, fertilisation techniques involving high resolution, digitally mapped soil data coupled with global positioning system (GPS) guided machinery (Hedley 2015). Therefore, further research needs to focus on deriving more simplistic approaches towards improving nutrient management strategies that can be implemented at minimal expense.

Predicting where major soil variations occur may help distinguish preferable sampling locations for agronomic testing, in turn, allowing soil data to be more representative of the overall management zone (e.g., paddock). Accordingly, this may result in the determination of fertiliser requirements to become more accurate; thus, improving fertiliser use efficiency. Previous studies have highlighted that in certain landscapes throughout eastern NSW, soil profiles between major alluvial units tend to vary (characteristically) along a gradient of increasing landform age, inferred from the height of the alluvial unit above the existing channel (Figure 1.1) (Walker & Coventry 1976, Walker 1989).

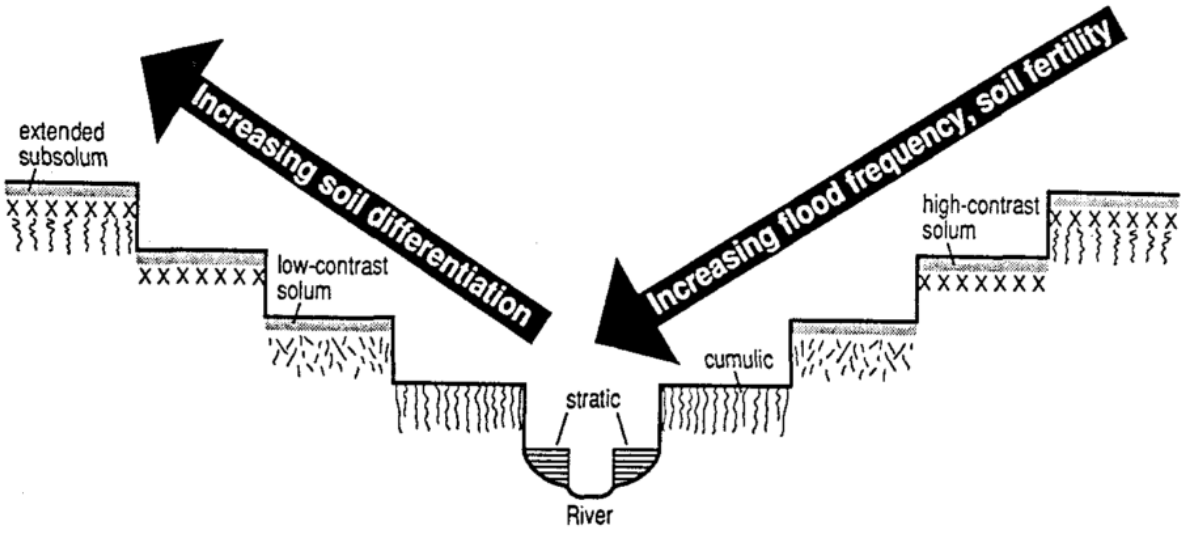

Figure 1.1. Common sequence of soil profile stages present within stepped alluvial landscapes found throughout eastern NSW (Figure from Walker 1989).

It is agreed among researchers that these variations reflect differential extents of soil profile development, with time acting as the primary differentiating factor (Brewer & Walker 1969, Warner 1972, Walker & Coventry 1976). Briefly, profile development can be explained as a progressive change in the initial sedimentological organisation of parent material into soil profiles displaying recognisable traits (Walker 1989). Profile development is fundamentally caused by outcomes of weathering processes, effectively altering both the physical and

chemical characteristics of the alluvial soils (Brewer & Walker 1969, Green & Walker 1976). Profile development may, therefore, be recognised to have an influence on some of the key factors known to be strongly associated with phosphorus dynamics. Hence, it is possible that P dynamics may change in a somewhat predictable pattern in alluvial landscapes consisting of an assemblage of stepped terrace and modern floodplain units. Despite this, research linking these two major areas of research remains largely unexplored.

1.2. Study area

The Manning River, residing on the mid-north coast of NSW, is a significant river system in both magnitude and as an important resource (Hughes & Watkins 2011). During the past century, alluvial landscapes adjacent to the Manning River have been extensively utilised for agriculture (Hughes & Watkins 2011). Among these agricultural operations, it is recognised that intensified dairy farms have perhaps the greatest potential to contribute considerable amounts of P contamination if farm-scale nutrient balances are not managed effectively (Gourley *et al.* 2012, McDowell 2012, Smith *et al.* 2013).

This study investigates a major intensified dairy operation situated on an alluvial landscape associated with the Manning River in order to determine whether concerning losses of P are likely to be occurring. A case study approach has been adopted to allow for the study to be made achievable given limited the time and resources available. In addition, this particular site was chosen based on the morphological features present within the landscape, that is, the stepped assemblages of modern floodplain and terrace units; thus, enabling findings of this study to be applicable to other landscapes with similar morphological characteristics.

A number of other studies relating to patterns of soil profile variation within stepped alluvial landscapes have been carried out at locations along the eastern NSW border, ranging from as far south as Nowra (Shoalhaven River) through to as far north as Kempsey (Macleay River) (Walker & Coventry 1976). Landscapes of this particular (geo)morphological nature associated with the Manning River, however, have received little attention in this area of research.

1.3. Project scope

The primary purpose of the study was to identify how P dynamics may vary within alluvial landscapes, the implications this may have for P management strategies, and the environmental consequences if left unaccounted for. Therefore, the research question underpinning this study was:

"Are there variations in soil properties between adjacent alluvial units of variable height above the existing channel and if so, do they affect P dynamics?"

Two key hypotheses were established in order to address the research question:

(1) Grain size, mineralogy, and oxalate extractable Al (Al_{ox}) and Fe (Fe_{ox}) contents in soil profiles will vary in a pattern similar to those described in previous studies.
(2) P dynamics will vary between locations of varying height above the existing channel.

An extension of this is whether the findings can be used to improve nutrient management strategies (e.g., fertiliser use efficiency) in order to improve the sustainability of farm practices. Grain size, mineralogy and Al_{ox} and Fe_{ox} were chosen as key soil properties to compare, as they are the least likely to be significantly altered by landuse; thus, enabling results to be applicable to other landscapes, irrespective of landuse practices.

1.3.1 Study objectives

Four principle objectives were set for this study:

(1) Identify the variations in key soil properties between major alluvial units of varying height and whether or not variations display a pattern similar to those described in previous studies.
(2) Assess the influence that investigated soil profile variations have on P dynamics.
(3) Determine the possible implications of findings for improving nutrient management strategies for the study site and perhaps other farms set in similar landscapes.
(4) Assess the sustainability of the farm under study and risk of P losses likely to occur as a result of not taking soil profile variations between alluvial units into account.

1.4. Significance of this research

Research presented here has the potential to provide additional knowledge required to enhance the sustainability of agricultural enterprises operating in alluvial settings through improving nutrient (P) management strategies. Consequently, this may result in reducing the overall risk of P losses to occur. Implications for findings may be applicable to other agricultural operations set within similar landscapes, particularly those previously identified throughout eastern NSW (Walker & Coventry 1976).

This research might also provide impetus for improving methods of agronomic testing as well as farm (e.g., paddock) design, which account for inherent soil profile variations with both depth and location in these types of landscapes. Furthermore, through incorporating P dynamics, findings from the study may be used to build upon previous work carried out in alluvial landscapes commonly found throughout eastern Australia; i.e., those consisting of a stepped assemblage of terrace and modern floodplain alluvial units.

1.5. Thesis structure

This introductory chapter (Chapter 1) has established the context of the study and provided a general background of the research topics of fundamental importance, whilst highlighting the key knowledge gaps of interest throughout. Moreover, the principal aims of the study and significance of the expected outcomes have also been outlined.

Chapter two (2) presents a review of the literature, providing a further insight into the current scientific understanding of the areas under study; including, phosphorus and the Australian dairy industry (Section one) and soil variations within alluvial landscapes common in eastern NSW (Section two). Chapter three (3) outlines the materials and methods of the present study, including a brief overview of the study area as well as a description of the experimental design and how the data was analysed. Chapter four (4) presents the results of the present study. Chapter five (5) provides an interpretation of the results and discusses the major findings accordingly. Chapter six (6) summarises the major findings of the study and highlights key areas of research requiring further work. In addition, a number of recommendations specific to the study area as to how farm sustainability could be improved are given based on the findings of the study.

CHAPTER 2

Literature Review

In this chapter, a review of the literature is presented with consideration of the importance of understanding farm-scale phosphorus (P) dynamics in alluvial landscapes, and how this knowledge can be used to address a range of key environmental issues pertaining to the Australian dairy industry.

The review of the literature has been separated into two major sections. Section one provides a general context of the study by outlining a range of key issues in the Australian dairy industry with a focus on environmental concerns. In addition, a general description of the role and importance of P in dairy farms is given, followed by an overview of various forms and reactions of P in soils. Current and potential future efforts to reduce the environmental impacts of the Australian dairy industry are also presented. Section two of the literature review consists of an overview of common alluvial landforms and associated variable soil types relative to the study area.

Section One: Phosphorus and the Australian Dairy Industry

2.1. Current issues in the Australian dairy industry

2.1.1. Intensification

Dairy farms throughout Australia are continuing to intensify (Gourley *et al.* 2007b, Burkitt 2014). Over the past few decades, a growing number of dairy farms have been increasing stocking rates as well as boosting pasture production levels in order to obtain higher milk yields per animal, per unit area (Figure 2.1) (Gourley 2004, Gourley *et al.* 2007a, Smith *et al.* 2013, Burkitt 2014). In addition, the average total farm area for Australian dairies has also increased (Figure 2.1). The primary impetus for intensification fundamentally pertains to financial gains (Gourley & Ridley 2005, Bramley *et al.* 2011, Smith *et al.* 2013).

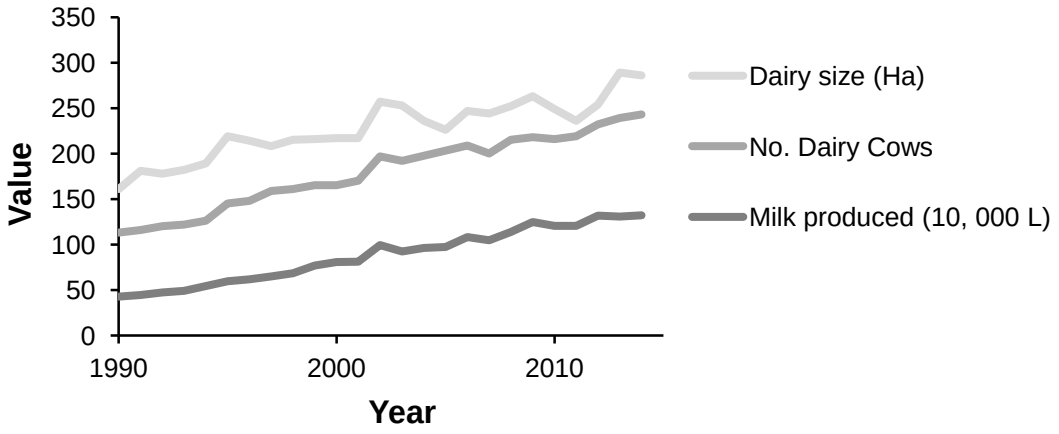

Figure 2.1. Annually averaged Australian dairy farm data derived from annual surveys conducted by the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES). See Appendix XV for raw input data.

Intensified dairy operations require much greater quantities of supplementary feeds and fertilisers in order to obtain the desired levels of pasture production and milk yields (Gourley 2004). This, however, often creates an imbalance between farm nutrient inputs and outputs (Sharpley *et al.* 2001, Gourley *et al.* 2007a, Burkitt 2014). Farm-scale nutrient imbalances can increase the risk of nutrients being lost to the environment (Holford *et al.* 1997, Gourley *et al.* 2010).

Large, intensified dairy operations are now very common throughout Australia, and as a result, environmental concerns are mounting (Gourley & Ridley 2005). Consequently, the Australian dairy industry has gained strong political and public attention in recent years (Gourley *et al.* 2010, Smith *et al.* 2013). Nonetheless, preventing further intensification of the dairy industry is not a viable option, as Australian dairies must maintain production levels which satisfy the growing population and increasing demand for dairy products (Fardeau 1995, Gourley *et al.* 2010, Weaver & Wong 2011). Therefore, efforts towards minimising the potential environmental impacts of intensified dairy farms operating in Australia continue to be widely encouraged (Gourley *et al.* 2007a, Smith *et al.* 2013, Simpson *et al.* 2014).

2.1.2. Dominance of large scale dairies

Small scale dairy farms are finding it increasingly more difficult to compete with larger, more intensified operations; hence, the total number of dairy farms in Australia has decreased over the past few decades (Figure 2.2) (Gourley 2004, Gourley *et al.* 2007a, ABARES 2015). As this continues to occur, the Australian dairy industry will undergo a fundamental shift into having fewer and larger dairy farms operating as intensified systems (Gourley & Ridley 2005, Smith *et al.* 2013). This is an important environmental concern as it would mean nutrient pollution generated by dairy farms would become much more concentrated, whereas in the past, these sources of pollution would have contained much lower levels of nutrients and have been much more spread out, allowing for better natural dilution of pollutants (Gourley & Ridley 2005).

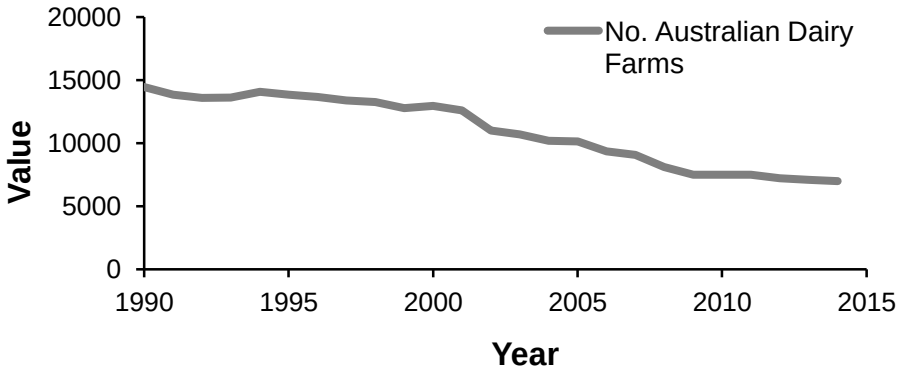

Figure 2.2. Number of Australian dairy farms recorded in an annual survey conducted by ABARES. See Appendix XV for raw input data.

2.1.3. Fertiliser use (in)efficiencies

Inputs of additional nutrients by means of fertilisation are often essential for maximising pasture production, especially in Australia, due to the widespread characteristically nutrient poor soils (Nash & Halliwell 1999). Fertiliser is commonly applied at rates in excess of what the receiving plant or crop requires for maximum production and thus remains a relatively inefficient practice (Gourley & Ridley 2005, Gourley *et al.* 2010). Furthermore, applications of recycled waste (e.g., manure and effluent) are generally restricted to a small portion of the overall farm area (i.e., close to milking sheds and storage facilities) as it is often not economically viable to expand the area to which the waste is applied (Whalen & Chang 2001, Gourley 2004).

Gourley *et al*. (2010) revealed that for 40 conventional dairy farms across Australia, 80% of paddocks utilised for pasture production had soil phosphorus (P) levels above the agronomic optimum (Gourley *et al.* 2010). Excess inputs of P may lead to significant accumulation in the soil, which consequently increases the risk of P being lost to the environment (Ige *et al.* 2005, von Wandruszka 2006, Gourley *et al.* 2007b, Dougherty *et al.* 2008, Mihailescu *et al.* 2014). Therefore, inefficient management of P fertilisers is recognised as one of the largest contributors of nutrient contamination in waterways; despite the fact that over fertilisation is often easily avoidable when the correct knowledge of nutrient requirements is available (Gourley & Ridley 2005).

2.2. Phosphorus: common inputs and importance within dairy farm systems

Phosphorus (P) is an essential element to life and its presence in natural ecosystems is nothing short of ubiquitous (Pierzynski *et al.* 2005, Stewart *et al.* 2005, Schoumans *et al.* 2015). Thus, P is absolutely critical for plant and animal growth/maintenance and is, therefore, one of the most important (macro)nutrients required for agricultural production (Correll 1998, Raghothama 2005, Satter *et al.* 2005, Worsfold *et al.* 2005, Gilbert 2009, Simpson *et al.* 2014). In order to maintain a financially sustainable dairy enterprise, it is essential that lactating dairy cows have sufficient levels of P in their diet, as P is required for not only maintenance and growth, but also pregnancy and milk production (Satter *et al.* 2005). The primary sources of dietary P in dairy farm systems are derived from grazed

pastures and supplementary feeds (Gourley *et al.* 2007a, Gourley *et al.* 2012). A large proportion of P in pastures is generally derived from P contained in chemical or organic fertilisers added to the soil (Simpson *et al.* 2014).

2.2.1 Chemical (inorganic) fertiliser

Practically all chemical P fertilisers are manufactured from naturally occurring phosphate-rich rock deposits which are mined in many countries throughout the world, including Australia (Figure 2.3) (Withers *et al.* 2005, Cooper *et al.* 2011, Vaccari & Strigul 2011, Schoumans *et al.* 2015). Phosphate-rich rock reserves are considered a non-renewable resource, and the availability, as well as quality of the material being mined is expected to decline (Cordell *et al.* 2009, Cooper *et al.* 2011, Achat *et al.* 2014). Researchers have estimated that global peak P production will have already been reached by 2030, and that the remainder of the reserve base will be largely exhausted by the end of the 21st century (Stewart *et al.* 2005, Cordell *et al.* 2009). Consequently, chemical fertilisers have risen in price over the past few decades and will continue to increase, simply as a result of supply and demand (Cordell *et al.* 2009, Weaver & Wong 2011, Peirce *et al.* 2013). Once again, this highlights the importance of improving P fertiliser use efficiency in contemporary agricultural operations.

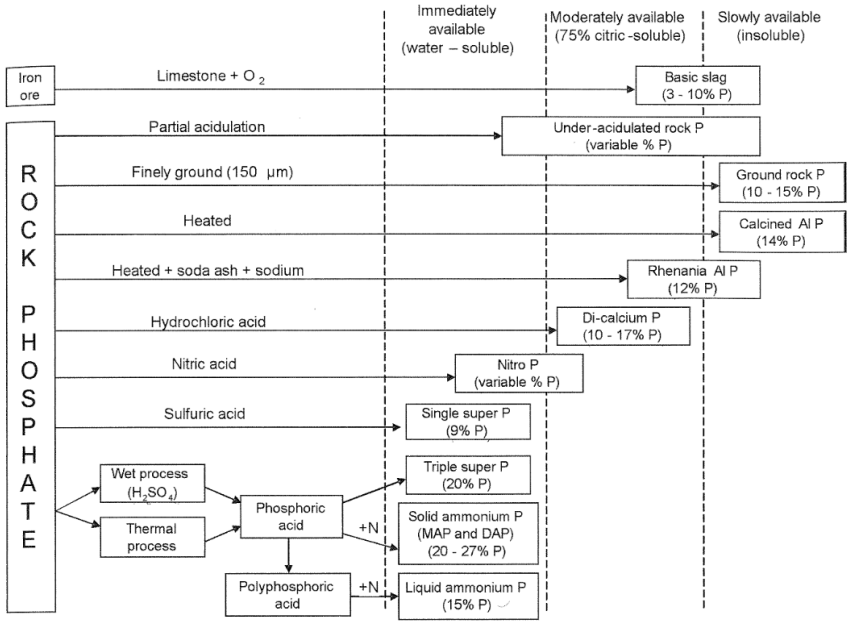

Figure 2.3. Inorganic P fertilisers, their manufacture and relative plant availabilities (Figure from Withers *et al*. 2005, pg. 785).

Typical forms of chemical P fertiliser used on Australian dairy farms largely consist of the common range of commercial superphosphates, for example, triple superphosphate (TSP) (Gourley 2004, Achat *et al.* 2014). These fertilisers are generally applied once a year just before autumn growth and are most commonly spread at a single rate within paddocks (Gourley 2004).

2.2.2. Organic fertiliser

Organic forms of P fertiliser are frequently used for agricultural production, as they are seen as a much cheaper, sustainable alternative to chemical fertilisers (Westerman & Bicudo 2005, Peirce *et al.* 2013). This is because organic P fertilisers are essentially made from recycled by-products (i.e., waste) of agricultural, industrial or municipal production(s) (Westerman & Bicudo 2005). For example, animal manure derived from poultry, swine, and dairy and beef productions (Sharpley & Moyer 2000). Moreover, the wider adoption of organic fertiliser use is reducing phosphate rock exploitation, which may see current reserves last a lot longer than what has been predicted (Achat *et al.* 2014). Irrespective of the benefits of organic waste utilisation (e.g., as fertiliser), there still remains a number of barriers which are limiting the wider adoption of organic fertiliser use (Figure 2.4) (Westerman & Bicudo 2005).

Figure 2.4. Challenges expected when transitioning from production (inner circle) to utilisation (outer circle) of organic wastes (Figure from Westerman & Bicudo 2005).

2.2.3. Supplementary feed

Supplementary feeding is widely recognised as an effective approach to achieving higher milk yields, per animal (Gourley *et al.* 2007a). In the year 2004-05, 91% of all dairy farms in Australia used imported supplementary feeds (Gourley *et al.* 2007b). Moreover, in the year 2009, 52% of Australian dairy farms fed cows a moderate-to-high rate of supplementary feeds in the order of greater than 1.0 t/cow/year (Gourley *et al.* 2007b, Bramley *et al.* 2011). It has also recently been reported that, on average, supplementary feeds account for about 40% of the typical Australian dairy cow diet (McDowell & Nash 2012). Therefore, supplementary feeds are heavily relied upon and may now be considered an essential dietary component for milking cows in Australian dairy farms (Gourley 2004, Gourley *et al.* 2007a).

Common types of supplementary feeds generally include grain mixes, pellets, hay, and silage (Gourley 2004, Bramley *et al.* 2011). Many of these supplementary feeds are made up of plant-derived ingredients and often contain higher concentrations of P than are found in the grazed pastures (Dougherty *et al.* 2008). As these feeds are predominantly imported from off-farm sources, the P which they contain must be considered as 'additions' to the overall farm P-cycle (Simpson *et al.* 2011). Thus, the increasing reliance on supplementary feeding to sustain intensified dairy operations is consequently resulting in more widespread farm-scale nutrient imbalances (McDowell & Nash 2012).

2.3. Cycling of phosphorus in dairy farm systems

The cycling of P in soils is a dynamic and complex phenomenon which is largely controlled by the nature of inorganic and organic solid phases present, the forms and degree of biological activity, the overall chemistry of the soil solution, and a range of environmental factors (e.g., rainfall and temperature) (Pierzynski *et al.* 2005). In dairy farm systems, the cycling of P is strongly influenced by relative amounts of inputs and outputs, which in turn, play a major role in the overall P sources and sinks (Figure 2.5) (Simpson *et al.* 2011).

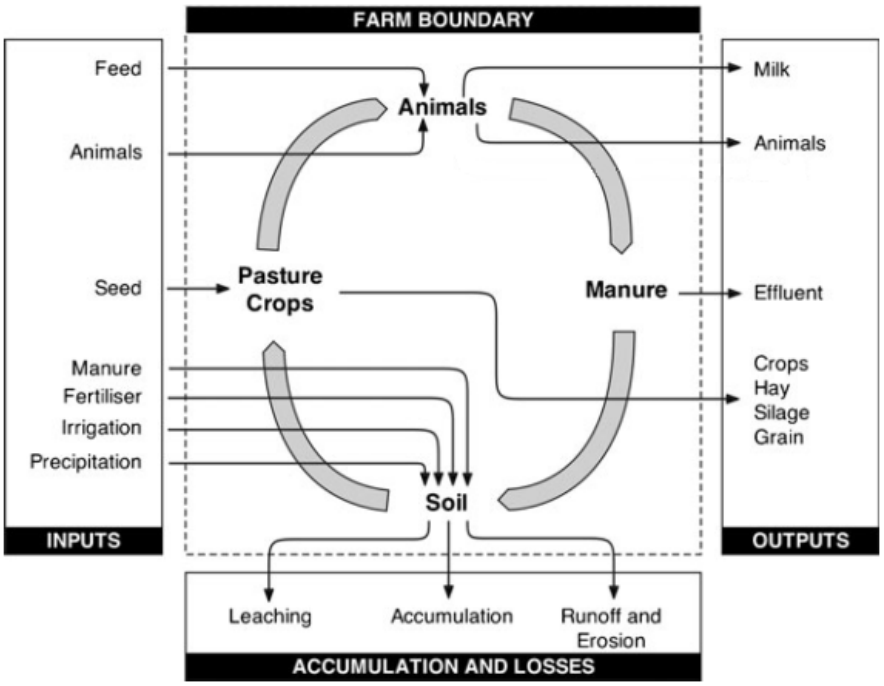

Figure 2.5. Generic framework of phosphorus cycling in common dairy farm systems (adapted from Weaver & Wong 2011).

The major knowledge gaps pertaining to P cycling in dairy farm systems relate to how P is influenced by different soil properties in response to varying treatments and or landuse practices (Wang *et al.* 2015). Therefore, further research is required to identify phosphorus loss relationships among varying soil types and how different soil properties may influence transport mechanisms.

2.3.1. Nutrient recycling

The benefits of nutrient recycling have been valued by farmers for centuries (Stewart *et al.* 2005). Within most dairy farm systems, nutrients are recycled through collection and subsequent application of manure and urine (i.e., 'animal P') directly to soils (Figure 2.5) (Gourley 2004). Additional 'incidental' recycling of nutrients also occurs in these systems through natural excretions made by livestock whilst grazing in the paddock (Gourley 2004). Nutrient recycling practices (although effective) are often inefficient, which is generally attributable to a lack of investment into effective collection, storage and redistribution techniques, as well as inherently variable P concentrations of organic wastes (Gourley 2004, Gourley *et al.* 2012). These investments, however, are often hard to justify, particularly when the majority of nutrient recycling practices are merely considered a convenient means

of disposal. Further research may provide further impetus for wider adoption of nutrient recycling practices as a primary fertilisation technique, such as, investigating the risks associated with utilising (i.e., grazing) effluent treated pastures for milk production.

2.4. Forms and reactions of phosphorus in soils

Soil P may exist in either organic or inorganic forms (Figure 2.6). The amounts of labile P contributed to the soil solution by organic and inorganic forms of P are largely controlled by mineralisation and sorption mechanisms, respectively (Yuan *et al.* 2007). Hence, these mechanisms primarily determine the movement of P in the soil matrix, whilst also influencing the P available for uptake by plants (i.e., biologically available P) (Schachtman *et al.* 1998).

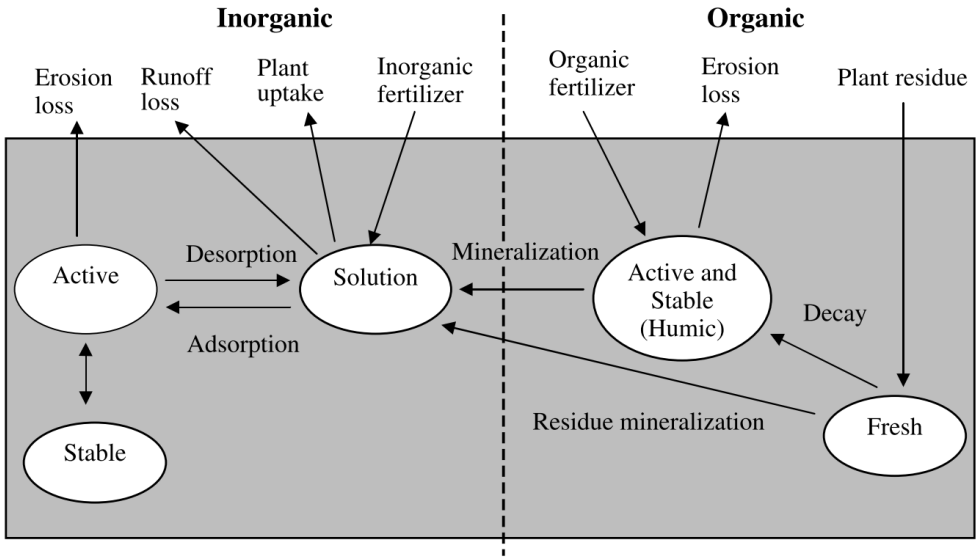

Figure 2.6. Forms and reactions of phosphorus in soils (Figure from Yuan et al. 2007).

2.4.1. Organic phosphorus

Organic P includes all P bonded in some way with carbon (Condron *et al.* 2005). Organic P typically makes up between 30% and 65% of the total soil P, but, in some cases it may be well above or below these limits (Condron *et al.* 2005). In grazed pasture systems (e.g., dairy farms), approximately 60-95% of P taken up by plants is returned to the soil as organic P on an annual basis, either directly in the form of litter and root residues or indirectly as animal excreta (Figures 2.5 and 2.6) (Condron *et al.* 2005).

Transformations of organic P in the soil play an important role in determining the overall biological availability of P (Nash & Halliwell 1999, Condron *et al.* 2005). Organic P may become biologically available (i.e., bioavailable) through mineralisation processes, which effectively release organically bound P into the soil solution (Figure 2.6) as soluble phosphate ions (Brady & Weil 1996, Condron *et al.* 2005). Organic P may also be immobilised via microbial breakdown when P is retained within the microbes biomass (Brady & Weil 1996). Environmental variables (e.g., precipitation and temperature) as well as landuse practices (e.g., irrigation and tillage) may influence the rates at which mineralisation occurs (Condron *et al.* 2005, Simmonds *et al.* 2015). Thus, total P concentrations contained in organic forms of fertiliser (e.g., manure and effluent) do not necessarily reflect the amount of bioavailable P once applied to a soil (Sharpley & Moyer 2000).

2.4.2. Inorganic phosphorus

Soil P content, particularly within the soil solution (Figure 2.6), is typically dominated by inorganic forms of P (Tan 2000, Abdala *et al.* 2015). Hence, plants tend to assimilate P primarily in inorganic forms (Schachtman *et al.* 1998, Raghothama 2005).

Inorganic P in soil may be present in several various forms of phosphate ions, such as: $H_2PO_4^-$, HPO_4^{2-}, and PO_4^{3-} (Tan 2000). The ratio of these ions in the soil solution is largely controlled by pH (Schachtman *et al.* 1998); this can be illustrated using the pK_a values of orthophosphoric acid (H_3PO_4) (Tan 2000):

$$H_3PO_4 \rightleftharpoons H^+ + H_2PO_4^- \qquad pK_{a1} = 2.17$$

$$H_2PO_4^- \rightleftharpoons H^+ + HPO_4^{2-} \qquad pK_{a2} = 7.31$$

$$HPO_4^{2-} \rightleftharpoons H^+ + PO_4^{3-} \qquad pK_{a3} = 12.36$$

The pK_a value is defined as the pH at which a compound dissociates yielding equal concentrations of anions and the original material (Tan 2000).

2.4.3. Phosphorus sorption

Sorption and desorption reactions are widely recognised as one of the most fundamental aspects of the chemical behaviour of P in soil (Bache & Williams 1971, Ahmed *et al.* 2008). Sorption is a term used to describe the process whereby P ions are removed from the soil solution via two mechanisms: (1) *ad*sorption and (2) *ab*sorption (Withers & Jarvie 2008). Sorption may, therefore, be considered a two-step process, involving both surface *ad*sorption (or desorption) of phosphate ions in solution to (or from) the solid phase and diffusion into (or out of) the solid phase via *ab*sorption (or desorption) (Barrow & Shaw 1975, Withers & Jarvie 2008). Barrow & Shaw (1975) suggested that sorption processes may be seen to occur in three separate compartments ('A', 'B' and 'C'), shown by the following equilibrium reaction:

$$A \rightleftharpoons B \rightleftharpoons C$$

Compartment 'A' represents phosphate in solution and may be transferred to compartment 'B' via *ad*sorption processes, resulting in the phosphate ions becoming attached to the soil particle surface (Figure 2.7) (Barrow & Shaw 1975).

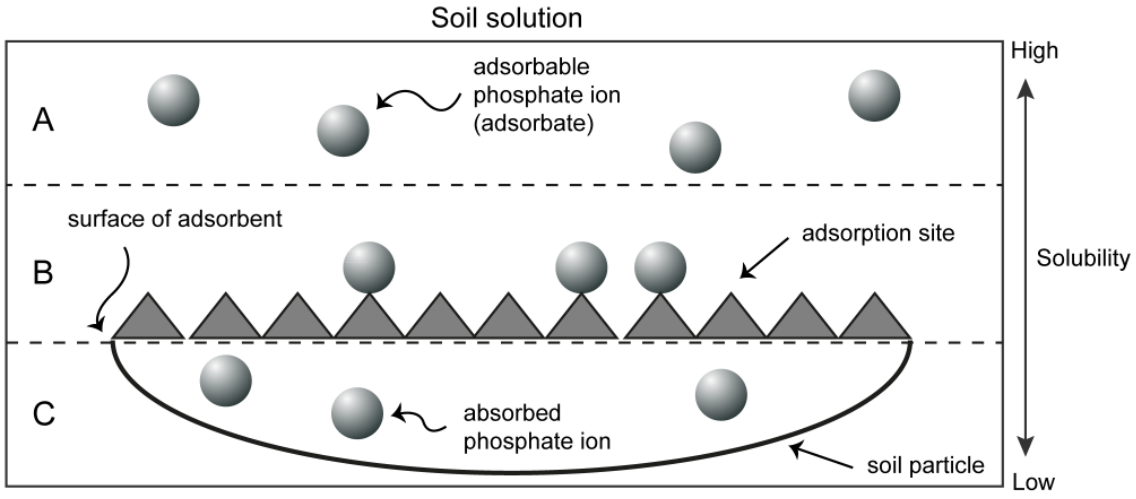

Figure 2.7. Diagrammatic illustration showing phosphate sorption in soils (derived from vanLoon & Duffy 2010, pg. 327 and Barrow & Shaw 1975).

*Ad*sorption is predominantly driven by ligand-exchange reactions (i.e., at Me-OH and Me-OH^{2+} sites), electrostatic attraction and displacement which is caused by anions competing for surface sites on soil particles (Vaananen *et al.* 2008, Withers & Jarvie 2008, Wu & Sansalone 2013). P may move from compartment 'B' to 'C' via *ab*sorption (i.e., diffusion)

processes, which may also be considered a 'precipitation reaction'; whereby over time, *ad*sorbed P penetrates the *ad*sorbent, making it increasingly insoluble and eventually rendering P completely unavailable for plant uptake (Figure 2.7) (Barrow & Shaw 1975, Brady & Weil 1996, Holford *et al.* 1997, Withers & Jarvie 2008, Wu & Sansalone 2013). However, this can be reversed as P may become available again through desorption processes (Barrow & Shaw 1975). When P is in either compartment B or C it is considered to be 'sorbed'. Therefore, sorption can be used as a collective term to describe the process whereby P is transferred from soil solution (liquid phase) to soil particles (solid phase) (Kerr *et al.* 2011).

2.5. Factors which may influence phosphorus sorption in soils

There are a range of known factors which may influence a soils ability to sorb P (Dougherty *et al.* 2010, Coad *et al.* 2014). Major factors known to play a role in P sorption include: aluminium and iron oxyhydroxide content, calcium carbonate content, pH, the nature and amount of clay constituents, organic matter, temperature, saturation of sorption complexes and moisture content (Dougherty *et al.* 2010, Janardhanan & Daroub 2010, Agudelo *et al.* 2011, Yaghi & Hartikainen 2013, Coad *et al.* 2014, Perez *et al.* 2014).

2.5.1. Aluminium and iron oxyhydroxide content

Phosphate sorption in soils is largely influenced by the presence of amorphous oxyhydroxides of aluminium (Al) and iron (Fe), especially when pH is < 7 (Gasparatos *et al.* 2006, Vaananen *et al.* 2008, Wang *et al.* 2010, Wang *et al.* 2015). Therefore, high naturally occurring concentrations of Al and Fe oxyhydroxides in soil often coincide with a high sorption capacity (Ahmed *et al.* 2008, Vaananen *et al.* 2008, Abdala *et al.* 2015). Concentrations of Al and Fe oxyhydroxides which are largely associated with sorption reactions are generally determined by the acid oxalate extractable method (Rayment & Lyons 2011).

In soil, Al and Fe oxyhydroxides occur as (short range order) crystalline structures and also as coatings on the internal and external surfaces of clay particles (Brady & Weil 1996). Al and Fe oxyhydroxides may sorb P by an exchange reaction, whereby singly coordinated hydroxyl groups at the sesquioxide surface are effectively replaced by phosphate ions (Figure 2.8) (Gasparatos *et al.* 2006, Wu & Sansalone 2013). Once P comes into contact

with the soil (e.g., via fertiliser application), these reactions can occur very rapidly and are, therefore, very effective at removing P from the soil solution (Barrow 1992).

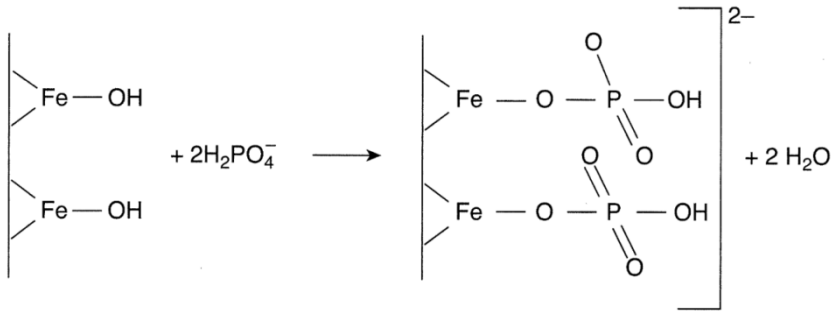

Figure 2.8. Specific adsorption of the phosphate ion onto an iron (Fe) oxide surface (Figure from vanLoon & Duffy 2010, pg. 433).

The reverse of this reaction may also occur under certain conditions (see below, section 2.5.3), resulting in the previously sorbed phosphate ion being released (desorbed) back into the soil solution (see above, section 2.4.3) (Yaghi & Hartikainen 2013).

2.5.2. Calcium carbonate content

Calcium (Ca) is a very important cation in soils, and commonly present in the form of calcium carbonate, especially in dryer zones (Tan 2000). Calcium carbonate ($CaCO_3$) content may play a significant role in controlling P sorption capacity under alkaline (i.e., pH > 7) conditions (see below, section 2.5.3) (Brady & Weil 1996, von Wandruszka 2006, Wang *et al.* 2015). In alkaline soils, highly soluble forms of P readily react with $CaCO_3$ to form less soluble (Ca-P) compounds (Brady & Weil 1996). Subsequent reactions may form sequential compounds with decreasing solubility and hence rendering them unavailable for plant uptake (Brady & Weil 1996).

2.5.3. Soil pH

Soil pH levels play a major part in controlling the solubility of the most common forms of sorbed P (i.e., Fe-P, Al-P and Ca-P), thereby, influencing the strength of attraction between P in the soil solution and potential sorption sites (Figure 2.9). Hence, pH may be considered the main determining factor in whether sorption or desorption processes will dominate which, in turn, has either a positive or negative (respectively) effect on the overall soil sorption capacity (Singh & Gilkes 1991, Gasparatos *et al.* 2006, Yaghi & Hartikainen 2013).

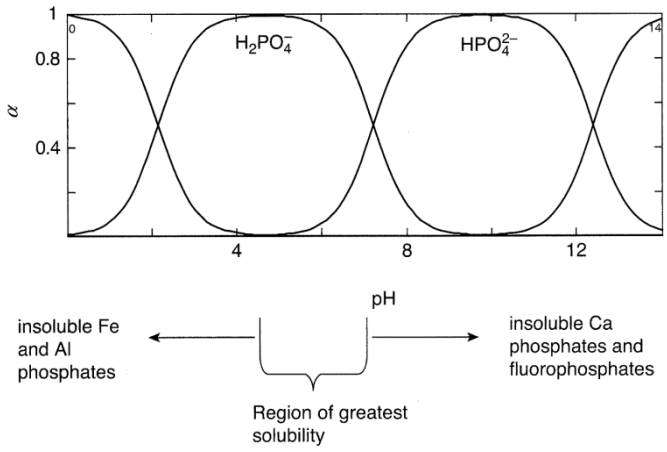

Figure 2.9. Relative solubility of sorbed P species (Al-P, Fe-P and Ca-P) depending on pH (Figure from vanLoon & Duffy 2010, pg. 330).

2.5.4. Nature and amount of clay constituents

Sorption capacity is often strongly correlated with clay content, where soils rich in clay tend to have higher P sorption capacities (Singh & Gilkes 1991, Gasparatos *et al.* 2006, Moody 2007). This is largely attributable to the fact that Al and Fe oxyhydroxides are predominantly present in the clay fraction of a soil (Singh & Gilkes 1991, Agudelo *et al.* 2011). In addition, the relatively small particle size of clays (e.g., < 2 µm) also plays a role, due to the large area of reactive surfaces enhancing the availability of potential sorption sites (Gasparatos *et al.* 2006, Withers & Jarvie 2008). Hence, clays with short range order crystalline structures (i.e., microcrystalline) will be able to sorb more P than well crystalline (macrocrystalline) clays (Singh & Gilkes 1991). Common types of clay minerals which have been associated with P sorption include: kaolinites, smectites, goethite, and gibbsite (Brady & Weil 1996, Withers & Jarvie 2008).

2.5.5. Organic matter

The literature suggests that the influence of organic matter (OM) on P sorption capacity is difficult to generalise, as various studies have revealed OM may effectively increase or decrease P sorption, quite often acting simultaneously (Haynes & Swift 1989, Iyamuremye *et al.* 1996, Janardhanan & Daroub 2010, Abdala *et al.* 2015). Whether the net influence of OM on the P sorption capacity of a given soil results in a positive (i.e., increase), negative (i.e., decrease) or neutral effect is largely dependent on the type of organic acid being

produced from the OM present, as well as the nature of the interactions occurring (Iyamuremye *et al.* 1996, Janardhanan & Daroub 2010).

There are two classes of interactions that explain the relationship between P sorption and OM - direct and indirect (Janardhanan & Daroub 2010, Abdala *et al.* 2015). Direct interactions have been associated with decreases in P sorption capacity, whereas indirect interactions have often been related to increases in P sorption capacity (Janardhanan & Daroub 2010). Indirect interactions between soil Al/Fe oxyhydroxides and OM may result from different organic acids being produced (e.g., malic and acetic acids) which inhibit crystallisation (Haynes & Swift 1989, Janardhanan & Daroub 2010). This, therefore, would result in a positive correlation between OM and the P sorption capacity due to Al/Fe oxyhydroxides with shorter, less established (short range order) crystalline structures having a higher capacity to sorb P than crystalline forms due to their larger surface area per unit volume (Janardhanan & Daroub 2010). Direct interactions may occur when acids produced by OM and P compete for sorption sites; thus, resulting in a negative correlation between OM and P sorption capacity (Haynes & Swift 1989, Brady & Weil 1996, Iyamuremye *et al.* 1996, Janardhanan & Daroub 2010, Simmonds *et al.* 2015).

2.5.6. Temperature

Research has shown that the overall sorption of phosphorus in soils may be influenced by temperature (Chen *et al.* 1982, Perez *et al.* 2014). The effects of temperature may vary among *ad*sorption, *ab*sorption and *de*sorption processes depending on soil characteristics, such as, pH (Barrow & Shaw 1975, Chen *et al.* 1982). Thus, variations in temperature may either result in an increase, or decrease in overall P sorption capacity of a soil (Barrow & Shaw 1975, Barrow 1992). It is worth noting, however, that the reported effects of temperature on P sorption may be conflicting throughout the literature as a result of inconsistent experimental methods (Barrow & Shaw 1975, Barrow 1992).

2.5.7. Saturation of sorption sites

The saturation of sorption complexes is a measure of the vacancy of sorption sites present in the soil (Vaananen *et al.* 2008, Perez *et al.* 2014). The P sorption capacity of any given soil is largely limited by the number of available sorption sites; therefore, the saturation of sorption complexes plays a significant role in determining the overall P sorption potential of a soil (Holford *et al.* 1997). In addition, P saturation has also been shown to be strongly

correlated with P desorption, such that as the saturation of sorption complexes increases the amount of P desorption increases (Sharpley *et al.* 2001).

2.5.8. Moisture content

Moisture content may influence a soils ability to sorb and desorb phosphorous (Brady & Weil 1996, Yaghi & Hartikainen 2013). A reduction in soil moisture content may sometimes increase the P sorption potential, as drying may cause OM molecules to condense, allowing previously obstructed Al/Fe oxyhydroxides to effectively sorb P (Haynes & Swift 1989). Drying of a soil, however, may also influence the crystallinity (i.e., size and order) of Al and Fe oxyhydroxides, thereby, reducing their overall surface area and hence ability to sorb P (Singh & Gilkes 1991, Worsfold *et al.* 2005, Janardhanan & Daroub 2010).

Desorption of P may be triggered by rapid wetting (i.e., rehydration) of the soil, as this may kill significant amounts of microbes as a result of osmotic shock and cell rupture (i.e., lysis), consequently, releasing more P back into the soil solution (Worsfold *et al.* 2005). Rehydration may also catalyse weathering mechanisms that cause soil aggregates to break down, effectively exposing P which was previously highly insoluble (absorbed) and thus potentially enabling it to be desorbed (Worsfold *et al.* 2005).

Furthermore, soil moisture can influence redox conditions which may have an effect on the solubility of sorbed P compounds (Pierzynski *et al.* 2005); for example, under anoxic conditions, Fe^{3+} is reduced to Fe^{2+}, consequently, rendering Fe-P more soluble (Wu & Sansalone 2013, Yaghi & Hartikainen 2013). This, therefore, lowers the overall retention of P and hence reduces the overall P sorption potential of the soil (Wu & Sansalone 2013, Yaghi & Hartikainen 2013).

2.6. The environmental importance of phosphorus sorption

P sorption is important as it plays a major role in minimising losses of excess P on farms which might otherwise lead to severe environmental problems (Simpson *et al.* 2011). In fact, sorption processes are so effective at removing P from solution that they are often used to treat contaminated waters (Brady & Weil 1996, Yaghi & Hartikainen 2013, Perez *et al.* 2014). Materials coated with, or naturally high in Fe and Al oxyhydroxides are sometimes

added to soils and or sediments in order to enhance P removal for water treatment purposes, for example, in constructed wetlands used for wastewater treatment (Brady & Weil 1996, Yaghi & Hartikainen 2013, Perez *et al.* 2014).

2.6.1. Consequential environmental impacts of phosphorus losses from agricultural systems

Losses of phosphorus through leaching, runoff and erosion can have detrimental effects on both ground and surface water quality (Sims *et al.* 1998, Nash & Halliwell 1999, McDowell & Nash 2012). Elevated levels of phosphorus in rivers may lead to *eutrophic*ation of surface waters (Figure 2.10), otherwise known as nutrient enrichment (Chittleborough & Cox 1999, Gourley & Ridley 2005, Burkitt 2014). Despite the fact that both nitrogen (N) and P are required to establish eutrophication, P is considered to be the key limiting element (Correll 1998, Barlow *et al.* 2003, Wu & Sansalone 2013, Yaghi & Hartikainen 2013).

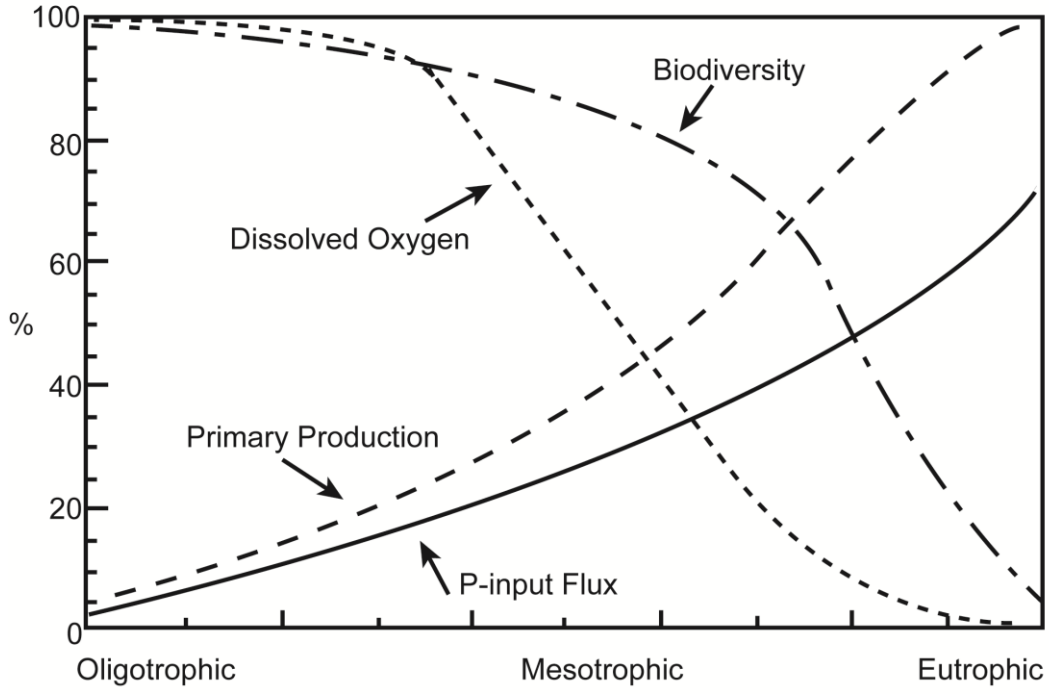

Figure 2.10. Conceptualisation of freshwater eutrophication **(**adapted from Correll 1998).

Eutrophication stimulates increased algal and aquatic weed growth (i.e., primary production; Figure 2.10), some of which may be toxic to humans, e.g., blue green algae (*Cyanobacteria*), thus, resulting in undesirable biodiversity changes (Figure 2.10) (Sharpley *et al.* 2001, Gourley & Ridley 2005, Burkitt 2014). In addition, when this material dies, it is

consumed by bacteria and subsequently decomposed via a metabolic process requiring dissolved oxygen (Sharpley *et al.* 2001, Burkitt 2014). Depletions in dissolved oxygen in water systems can cause fish and other aquatic organisms to suffer from asphyxia which may ultimately result in death (Burkitt 2014). Therefore, P contamination in surface waters may severely threaten the health and biodiversity of aquatic ecosystems and may also degrade the quality of publicly used water resources – a problem which is currently costing Australia millions of dollars annually in water treatment expenses (Chittleborough & Cox 1999, Burkitt 2014). As a result, it is important for future Australian funding and research to prioritise this particular issue, especially due to the increasing intensification of the dairy industry, which may result in more frequent cases of eutrophication (Smith *et al.* 2013, Burkitt 2014).

2.7. Determination and analysis of the phosphorus status of soils

The P status for any given soil may be assessed via interpretation of a range of determined parameters. These interpretations may be intended for either agronomic or environmental purposes (Wang *et al.* 2015). Hence, different parameters or tests may be more relevant for one purpose compared to another, or, may be useful for both (Khiari *et al.* 2000, Wang *et al.* 2015). A more detailed overview of the various agronomic and environmental soil P tests has been provided in Appendix XVII.

Agronomic P tests are predominantly used to estimate fertiliser requirements (Wang *et al.* 2015). Hence, soil tests conducted (often routinely) for agronomic purposes primarily include those which have been designed to indicate how much soil P is available for plant growth throughout the growing season (Wang *et al.* 2015). Thus, traditional methods of agronomic sample collection generally involve sampling the top 0-25 cm (depending on the type of pasture or crop) of the soil profile, where the majority of soil-nutrient-plant interactions are expected to occur (Sharpley *et al.* 1996, Rayment & Lyons 2011).

In contrast, environmental tests include a range of parameters which are able to be used to predict or determine potential losses of farm-P to the environment (Wang *et al.* 2015). These tests, however, generally lack uniformity and are not well suited to routine analysis (Pöthig *et al.* 2010).

Recent research efforts have focused on developing agronomic-type tests (e.g., phosphorus buffering index - PBI) that have additional environmental applications as well (Burkitt *et al.* 2002). For example, the PBI can be used to help determine more accurate estimations of P fertiliser requirements needed to achieve optimal yields (e.g., based on critical Colwell P concentrations; Table 2.1), in turn, minimising the risk of P accumulation in the soil and potential losses to the environment (Gourley *et al.* 2007a). In addition, PBI can be used to indicate a soils susceptibility to subsequent losses of P following fertiliser application, as it also provides a measure of the soils sorption capacity (Weaver & Wong 2011).

Table 2.1. Estimated critical Colwell P values for standard PBI categories generated from data derived from an Australian national data set (Gourley *et al.* 2007a; Table 2).

PBI range	PBI category	Critical Colwell P value (mg/kg) for mid-point of PBI category (range)[1]
< 15	Extremely low	23 (20 - 25)
15-35	Very very low	26 (24 - 27)
36-70	Very low	29 (27 - 31)
71-40	Low	34 (31 - 36)
141-280	Moderate	40 (36 - 44)
281-840	High	55 (44 - 64)
> 840	Very high	n/a[2]

[1]Values in parenthesis are critical Colwell P values for the lowest and highest PBI values within the respective PBI range.
[2]Insufficient data.

Furthermore, studies have also attempted to reveal the potential use of current agronomic tests for environmental purposes, namely, P loss risk assessment (Hooda 2000, Moody 2011, Wang *et al.* 2012, Wang *et al.* 2015). However, further research is needed to validate whether these approaches can be used in a range of settings consisting of varying soil characteristics (Dougherty *et al.* 2011). Nonetheless, developing more soil P tests which can be used to make both fertility and environmental risk based assessments remains a desirable prospect; particularly if these tests can be made universally applicable and included in the general suite of routine agronomic soil P tests (Moody 2011).

2.8. The movement of phosphorus in soil and major loss pathways

Phosphorus can be lost via three primary modes of transportation: (1) surface runoff (i.e., overland flows), (2) subsurface flows and (3) erosion (Figure 2.11) (Sharpley *et al.* 1996, Nash & Halliwell 1999, McDowell *et al.* 2004, McDowell 2012, Simpson *et al.* 2014).

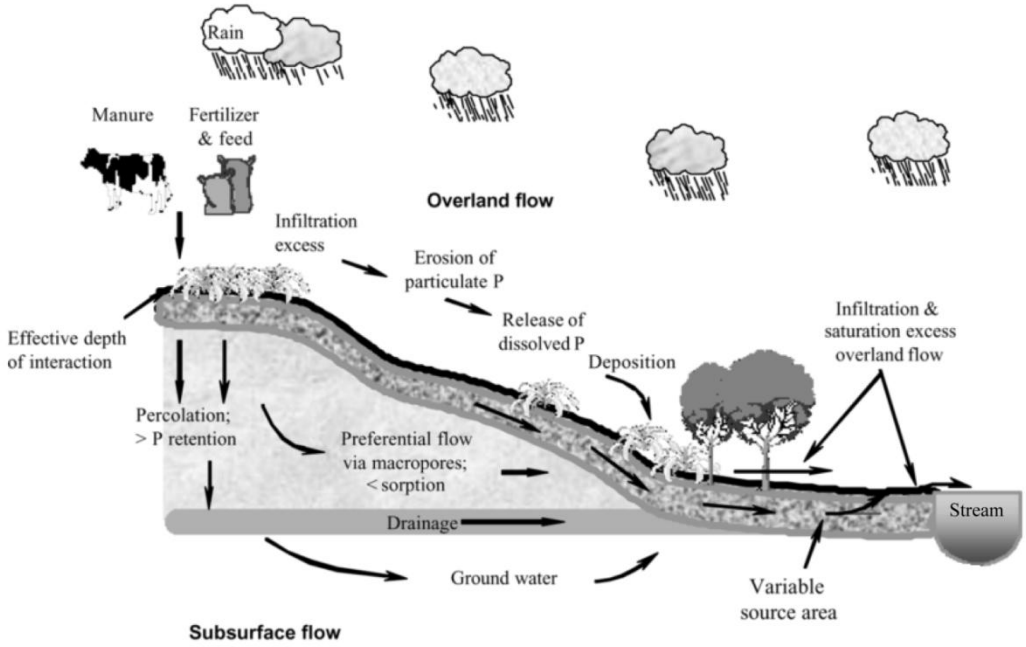

Figure 2.11. Major loss pathways of phosphorus in dairy farm systems (adapted from McDowell *et al.* 2004).

Surface runoff is generally perceived as the primary pathway of soil P losses to occur (Wang *et al.* 2012). In natural systems, P losses in surface runoff occur mainly in particulate form, but in intensified pasture production systems much more dissolved P losses are comparatively more likely (Gburek *et al.* 2005). Particulate P (PP) suspended in runoff may be removed from solution before reaching receiving water bodies by settling or filtration (Wu & Sansalone 2013), whereas dissolved P (DP) is often reduced via sorption and biological mechanisms (e.g., uptake by plants). However, P losses in surface runoff are largely induced by fast flow processes (e.g., during intense storm events), therefore, most of these removal mechanisms often have little influence on the total P lost (Gburek *et al.* 2005).

Subsurface pathways can also play an important role in the overall P losses from agricultural systems (Maguire & Sims 2002, Gburek *et al.* 2005, Wang *et al.* 2012). P has a high affinity for most soil types and therefore moves very slowly within the soil matrix (Ahmed *et al.* 2008). However, when concentrations of P are high and sorption potential is largely diminished, P may move more freely (Ahmed *et al.* 2008, Wang *et al.* 2012). Little field research has been conducted in order to predict the degree of P saturation required for leaching to occur, but, it has been indicated that it is possible for soils to leach P well before

the sorption capacity has been exhausted (Holford *et al.* 1997). In addition, factors such as grain size may also enhance P mobility within soils by providing preferential flow paths (Figure 2.11) for the soil solution (Sharpley *et al.* 2001, McDowell *et al.* 2004).

Despite erosion being a natural process, it is now considered a threat to the environment as a result of anthropogenic induced changes to the landscape, i.e., clearing of land coupled with accumulation of nutrients in soils (Gburek *et al.* 2005). Phosphorus attached to soil particles (i.e., PP) may be lost when soils are eroded by wind or surface runoff. Volumes of material and hence, concentrations of P lost via erosion are strongly influenced by the soil structure and particle size distribution (Gburek *et al.* 2005).

2.9. Solutions to minimise phosphorus losses in Australian dairy farms

Australian dairies must maintain production levels which correspond to the growing human population (Gourley *et al.* 2010, Weaver & Wong 2011). Therefore, simply restricting the levels of P inputs is not necessarily a viable option to reduce nutrient losses. Alternatively, P losses should be minimised through improved management strategies targeting better fertiliser-use efficiencies and implementation of P-loss mitigation strategies (Gourley & Ridley 2005, McDowell & Nash 2012).

2.9.1. Improved nutrient management through 'Precision Agriculture'

'Precision Agriculture' (PA) is a practice which has been recently developed and introduced on the back of rapid advances in geographical information science (GIS) and remote sensing (RS) technologies (Bouma *et al.* 1999, Hedley 2015). PA utilises these technologies to effectively map the distribution of nutrients and, in turn, nutrient requirements; therefore, enabling more efficient use of fertilisers (Hedley 2015). Once determining site-specific nutrient requirements, PA may be operationalised via advanced variable-rate fertilisation techniques achieved using GPS guided machinery and various sophisticated implements (Zhang *et al.* 2002, Hedley 2015). Thus, PA can be conceptualised as an approach towards reshaping agricultural enterprises into more sustainable operations (Zhang *et al.* 2002).

The adoption of PA in Australia is lagging in comparison to some other countries, such as, the USA (Zhang *et al.* 2002, Gourley *et al.* 2010). Economic benefits resulting from implementing these technologies are difficult to measure and it is, therefore, often assumed

that the costs involved may outweigh financial gains (at least in the short term) (Zhang *et al.* 2002). Hence, Australian farming enterprises are frequently reluctant to spend this money, regardless of the environmental benefits which may be achieved. Nonetheless, as dairy farms in Australia continue to intensify their operations, there will be a mounting need for PA, as it will play a major role in reducing losses of P as a result of inefficient fertiliser use (Gourley & Ridley 2005). In order to encourage the implementation of PA, the Australian Government may need to offer incentives, such as, subsidies – an approach which has proven to assist the adoption of these technologies in other countries (Barataud *et al.* 2014), but, is comparatively lacking in Australia (McDowell & Nash 2012).

2.9.2. P-loss mitigation strategies

Many P-loss mitigation strategies have already been developed to minimise nutrient losses on dairy farms in other parts of the world (Weaver & Wong 2011, Das *et al.* 2012). P-loss mitigation strategies may include practices such as: implementation of stream fencing, advanced irrigation and recycled waste management, application of P-sorptive materials, and optimised nutrient budgeting (Gourley *et al.* 2007b, McDowell & Nash 2012, Burkitt 2014).

2.9.3. More stringent regulations

Enforcing more stringent regulations may be required to convince farm managers to work harder to reduce farm P losses (Gourley & Ridley 2005, Barataud *et al.* 2014). However, the Government must have the capacity to enforce new regulations. In addition, to warrant stricter environmental regulations for dairy farms (e.g., nutrient losses) authorities must have sufficient evidence indicating the environment is under a certain level of threat. To obtain this information, scientific research must first be conducted.

2.9.4. Further research and education

As dairy farms continue to intensify in Australia it is important that research aims at identifying ways which farms can adjust their management strategies in order to ensure the protection of the environment. In addition, it is equally important that findings from this research are effectively passed on to farm managers, enabling them to make educated decisions and to apply improved management techniques first hand, thereby, improving their environmental performance (Gourley & Ridley 2005). This will allow farm managers

to obtain a better balance between production levels and environmental impacts – effectively improving overall sustainability. Moreover, it is important that farm managers have access to information regarding the financial benefits associated with improved nutrient management, providing them further impetus for adopting a more sustainable approach to fertiliser use (Gourley *et al.* 2007b).

Section 2: Common alluvial landscapes in eastern NSW

2.10. River terrace landscapes

Sequences of stepped alluvial terraces abutting a contemporary floodplain unit represent a common landform assemblage associated with many eastern NSW river systems (Warner 1972, Walker & Coventry 1976, Walker 1989, Cohen & Nanson 2008). Terraces found within these landscapes generally occur in a 'staircase' like sequence, whereby the age of the terrace is often inferred to become greater with increasing height of the terrace surface above the existing channel (Walker 1963, Brewer & Walker 1969, Walker 1989). The commencement of terrace studies in Australia have been relatively delayed compared to other parts of the world, with most of the research being conducted post 1950 (Warner 1972). Since then, studies have focused on a range of river systems along eastern NSW, ranging from as far south as the Shoalhaven River near Nowra to as far north as Macleay River near Kempsey (Walker & Coventry 1976). Very little work in this particular area of research, however, has been carried out on landscapes within the Manning River catchment; at least not studies which have focused on investigating variations in soil characteristics within terrace sequences (Walker & Coventry 1976).

Before commencing a discussion of how terraced landscapes form, firstly, it is important to establish what the term 'terrace' exactly refers to in this text. Throughout the literature there is a range of definitions associated with what constitutes an alluvial 'terrace', as well as the various forms in which they may exist (Leopold *et al.* 1964, Warner 1972, Summerfield 1991). One of the more widely used definitions is that derived from Leopold *et al.* (1964); where, the term *terrace* simply refers to an 'abandoned floodplain' (Warner 1972). That is, former floodplain units which are no longer associated with the current hydrological regime (Leopold *et al.* 1964, Warner 1972). This particular example is often considered applicable

to most studies, and is most commonly recognised as a useful 'working definition' (Warner 1972). Another suitable definition to be used in addition to the former is one originally proposed by Walker (1962c), who describes the term *terrace* as a 'sedimentary structure built by stream aggradation and having nearly plane surfaces'. The latter example is particularly useful in situations where the alluvial unit in question has an associated uncertainty attached as to whether the floodplain has in fact been abandoned.

There has been an abundance of research conducted and associated theories formulated regarding the formation of river terrace landscapes, some dating back over a century (Warner 1972). Patterns displayed in landscapes composed of stored sediment throughout eastern NSW (and many other parts of the world) are widely thought to have occurred primarily as a function of tectonic (in)stability, climate change and eustatic alterations throughout time (Cohen & Nanson 2008). Collectively (or in some cases individually), these factors may cause a shift in the hydrological regime, resulting in one of three types of floodplain abandonment, as shown in Figure 2.12 (Warner 1972).

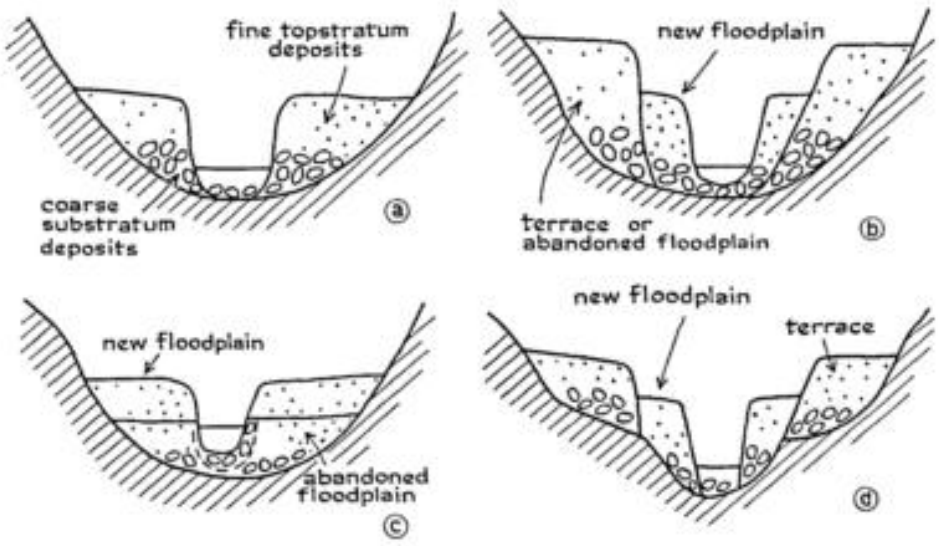

Figure 2.12. Types of terrace abandonment and associated morphological outcomes; where, 'a' is the 'original' floodplain and channel, 'b' is an inset terrace, 'c' is a buried terrace and 'd' is an incised terrace (adapted from Warner 1972).

Climatic change, sometimes accompanied by movements in sea level (i.e., eustatic changes), may alter a rivers discharge levels which, in turn, may affect the landscape surrounding its natural path (Walker 1970, Warner 1972, Walker 1989, Summerfield 1991, Cohen & Nanson 2008). In fact, major past climatic changes, such as that of the late

Quaternary, are often reflected by remaining evidence of floodplain adjustment displayed by preserved terraces (Warner 1972, Cohen & Nanson 2008). For example, long-term reductions in rainfall may bring a fall in discharge; this may result in established floodplains to be 'abandoned', and a new floodplain inset below the level of the old one would begin to establish itself via subsequent aggradation (Figure 2.12 (b)) (Walker 1962b, Warner 1972).

Conversely, climate change may also increase discharge levels, resulting in significant downcutting of the landscape (Walker 1962b, Leopold *et al.* 1964, Warner 1972). In this situation, downcutting followed by subsequent aggradation may effectively bury or, in much more extreme cases, completely replace the original floodplain (Figure 2.12 (c)) (Warner 1972, Nanson 1986). In addition, renewed erosion of the bedrock floor (sometimes) accompanied by uplift to produce a downcutting effect may result in an 'incised terrace' being formed (Figure 2.12 (d)) (Warner 1972, Summerfield 1991).

The way in which a given landscape responds to changes brought about by climate change is dependent on many factors, such as: valley confinement, nature of valley floor trough, nature of stream load, bedrock material, vegetation, and perhaps others (Leopold *et al.* 1964, Summerfield 1991, Warner 1992, Cohen & Nanson 2008). Therefore, different types of terrace abandonment may occur along different parts of a river system, depending on the diversity and nature of landscape settings through which it flows (Warner 1972, Warner 1992). For instance, in river systems on the north coast of NSW, terraces have been found to be incised in the upper valleys, inset in the lower middle reaches and buried in the lower parts of the systems (Warner 1972, Warner 1992, Cohen & Nanson 2008).

Sequences of stepped terraces are most commonly said to derive from repeated inset and/or incised forms of abandonment (Figure 2.12); hence, in cases where these events are repeated over time, flights of stepped terraces can be formed (Warner 1972, Walker & Coventry 1976, Walker 1989). For example, contrasting climatic phases causing alternating episodes of downcutting of the stream followed by aggradation to effectively generate a succession of terraces with incremental heights above the surface of the existing channel (Walker 1962b, Walker 1970). Therefore, sequences of stepped terraces often reflect 'cyclic' (or episodic) processes of floodplain formation (Warner 1972, Walker & Coventry 1976, Walker 1989). For this reason, assemblages of stepped alluvial units are generally assumed to become greater in age with increasing height above the existing channel, an assumption which has been shown to be valid in a number of landscapes (Brewer & Walker

1969, Warner 1972, Walker 1989). Consequently, individual alluvial units within stepped assemblages are commonly assigned *'K(-cycle)'* values used to represent the unstable phase responsible for their formation (Walker 1962b). For example, K_0 generally denotes the currently active (i.e., present epicycle) alluvial unit (i.e., modern floodplain), and the most recently abandoned terrace is assigned K_1 (Walker 1962b, Walker 1970, Cohen & Nanson 2008). This terminology has been adopted by researchers and was originally proposed by Butler (1959), who defined a "K-cycle" as a cyclic form of landscape evolution with recurrent 'unstable' and 'stable' phases, denoted as K_u and K_s, respectively (Walker 1962a, Walker 1962b). The use of age-height correlations to derive single K-cycle values within sequences of alluvial units has, however, been proven to be invalid in some landscapes and should, therefore, be interpreted with caution (Warner 1972, Cohen & Nanson 2008). Despite this, various methods used to assess the relative degrees of soil profile development in terrace sequences have frequently shown that terrace age-height correlations can often represent successions of major K-cycles (Brewer & Walker 1969, Warner 1972, Walker & Coventry 1976).

2.11. Patterns of soil profile development in stepped terrace landscapes

Soil profile development can be explained as a progressive change in the initial sedimentological organisation of the parent material into soil profiles which display signs of easily recognisable horizons, intense leaching, clay translocation and chemical weathering (Walker 1989). In order for deposited alluvium to commence profile development, the alluvial unit must be left free from disturbances from fluvial interactions, i.e., abandoned (Walker 1962b, Warner 1972). Consequently, differences in soil profiles are often found between stepped alluvial terraces as they have generally undergone different extents of profile development as a result of initiating at different points in time, as can be shown in Figure 2.13 (Brewer & Walker 1969, Walker 1989).

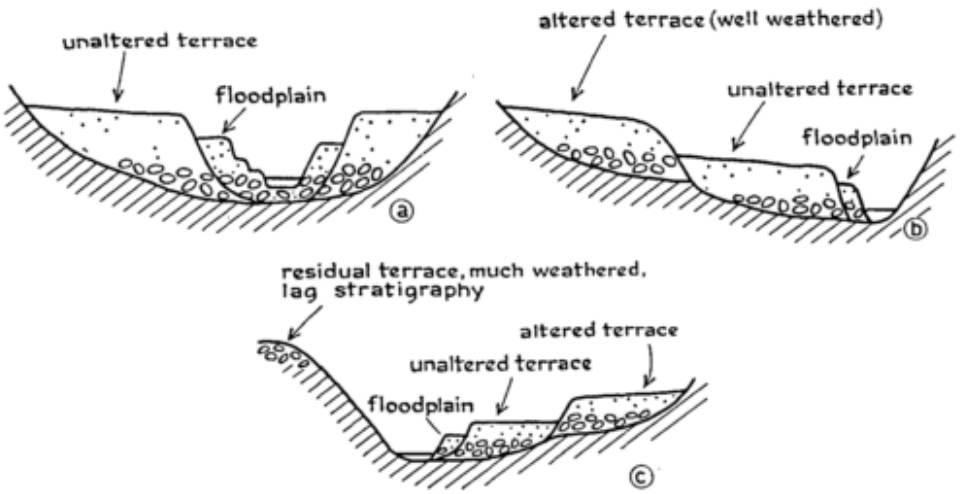

Figure 2.13. Types of river terraces based on degree of profile development following abandonment and stratigraphical preservation; where: 'a' is an inset unaltered terrace and floodplain; 'b' is an incised altered terrace, inset unaltered terrace and floodplain; and, 'c' shows incised residual and altered terraces, an inset unaltered terrace and a floodplain (adapted from Walker 1972).

Time is often considered the most influential factor controlling the outcome of profile development resulting in variations between terraces and is perhaps the easiest to quantify, as it is a factor which can be assumed to be constant (Brewer & Walker 1969, Warner 1972, Walker & Coventry 1976, Huggett 1998). Additionally, however, variations in the nature of material deposited and changes to the surrounding environment throughout time can also affect the outcome of soil development processes, making it difficult to accept that time alone is attributable for different extents of profile development between terraces (Brewer & Walker 1969, Warner 1972, Huggett 1998). For example, climatic conditions such as temperature and moisture (i.e., from precipitation) can affect the rate at which soil development occurs (Summerfield 1991). Similarly, the nature of material undergoing weathering processes may affect the rate of soil development but also may influence the nature of the resulting soil profile (Summerfield 1991). For these reasons, it may be sometimes difficult to draw conclusions regarding patterns of soil development in relation terrace age, as climatic changes and material deposited may sometimes be disproportionate throughout time. Nonetheless, the patterns of soil development which have been observed to occur in sequences of stepped alluvial terraces have been found to be similar across a number of landscapes in eastern NSW (Walker & Coventry 1976).

There are a number of soil properties which have been shown to vary between major alluvial units within stepped terrace landscapes (Walker 1962c, Brewer & Walker 1969, Warner 1972, Green & Walker 1976, Walker & Coventry 1976, Walker 1989). For example, differing expressions of biotic mixing and mineral weathering may result in variations in soil texture, grain size distribution, degrees of horizonation, porosity, and mineralogy (Green & Walker 1976, Walker 1989).

Differentiations in the nature and extent of weathering in a soil profile, a fundamental mechanism of profile development, is often regarded as the major cause of the variations in soil profiles found between alternating terraces (Brewer & Walker 1969, Cohen & Nanson 2008). However, it must be assumed that parent material across terraces is relatively similar in order for this explanation to be valid (Brewer & Walker 1969). In addition, this explanation also relies on the assumption that products of weathering were formed predominantly *in situ* (Brewer & Walker 1969, Green & Walker 1976). Accordingly, other factors such as inconsistent nature of alluvia as well as aeolian derived material (e.g., loess) being deposited over time have been raised as a major limitation of the assumption that weathering processes are primarily responsible for major soil profile variations between terraces of differing age (Warner 1972, Huggett 1998).

A typical progression of the common stages of profile development, including, *stratic*, *cumulic*, *low-contrast solum*, *high-contrast solum* and *extended subsolum* is shown in Figure 2.14. Radiocarbon dates have been determined for each of the general soil types found within various sequences of stepped alluvial units in eastern NSW, enabling approximate age ranges to be assigned to them, shown in Table 2.2 (Walker 1989). Stepped alluvial units which display (part of) the sequence of soil profile stages and associated soil types shown in Figure 2.14 and Table 2.2 are typically indicative of increasing soil development with landform (e.g., floodplain or terrace) age (Walker 1989).

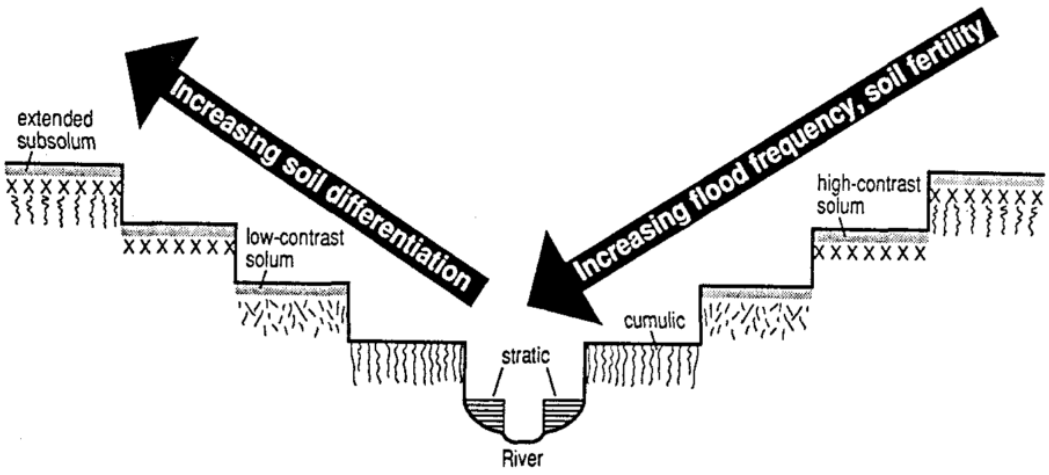

Figure 2.14. Common sequence of soil profile stages present within stepped alluvial landscapes found throughout eastern NSW (Figure from Walker 1989).

Table 2.2. A synthesis of information on stepped alluvial landscapes in eastern NSW, showing successive stages of soil profile development (as shown in Figure 2.14), (general) approximated soil type and age, and associated K-cycle(s) likely to be appropriate.

Stage of profile development	*Equivalent Soil Classification	Approximate Age (^{14}C)	Alluvial unit classification (tentative K-cycle)
Stratic	Entisol	contemporary	Low alluvial bench (K_u)
Cumulic	Inceptisol/Mollisol	< 2000 yr	Unaltered (K_0-K_1)
Low-contrast solum	Alfisol	5000-10,000 yr	Altered ($\geq K_1$)
High-contrast solum	Ultic Alfisol	20,000-30,000 yr	Altered ($\geq K_2$)
Extended subsolum	Utisol	Over-range	Ancient ($\geq K_2$)

Contents derived from Walker (1989), Walker & Coventry (1976) and Warner (1972)
*Soil classifications in accordance with USDA soil taxonomy based on authors interpretation of available data (Walker 1989).

2.12. Summary (sections one and two)

This literature review has indicated that despite there being a large body of research existing for the main areas of study outlined in sections one and two, researchers have made minimal attempt to establish some form of link between P dynamics and variable soil types common in stepped alluvial landscapes of eastern NSW. Forming this link may have implications for improving the sustainability of agricultural practices. Thus, further research is required to determine how soil profile variations in sequences of stepped alluvial assemblages affect soil P dynamics and, in turn, how may this influence farm management decisions.

CHAPTER 3

Materials and Methods

3.1. Introduction

The chapter begins with a brief overview of the study area and a description of the dairy farm under examination. The materials and methods used in the study are outlined, including: site selection, sample collection and sample analysis. In addition, a description of how the data was processed and analysed, including statistical methods, is provided.

3.2. Regional setting

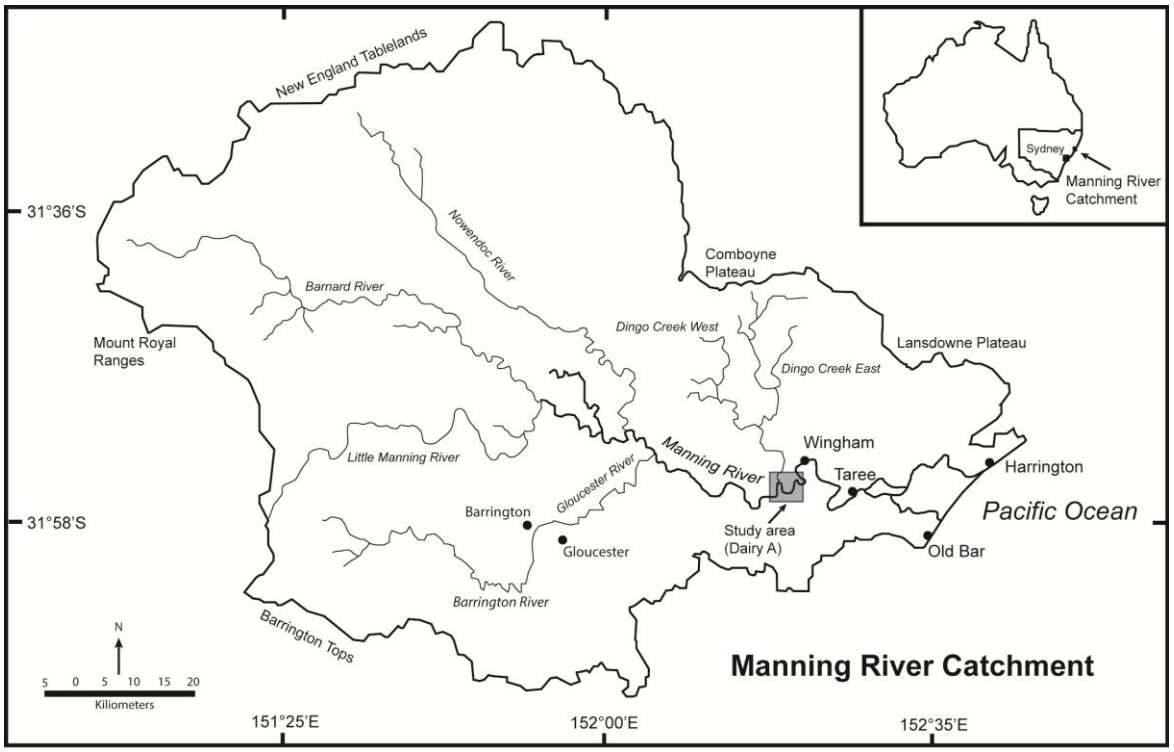

Figure 3.1. The study area

The study site was set on a dairy farm located on the Manning River on the mid-north coast of NSW, approximately 300 km NE of Sydney (Figure 3.1). The Manning River Catchment (Figure 3.1) has a total area of approximately 8400 km^2, characterised by steep forested highlands and dissected plateaus up to 1,500 m in elevation (predominantly occupied by National Parks and Forestry), intertwined with pockets of pastured slopes and plains utilised for agriculture (Nanson 1986, MCMC 1996, Hughes & Watkins 2011). Geology within the catchment consists of a complex network of faulted units, including: Silurian and Devonian slates, quartzites and acid volcanics; Carboniferous mudstones and lithic sandstones; Lower Permian shales and sandstones; and, some less prevalent Triassic sediments (MCMC 1996, Hughes & Watkins 2011, NSW OEH 2011). Granite and Cenozoic basalt plateaus are also present within the catchment, the latter capping a large proportion of the upper-catchment, thus, occurring across most of the catchments headwaters (Hughes & Watkins 2011, NSW OEH 2011).

3.3. Climate and hydrology

The study site experiences predominantly sub-humid temperate conditions, generating a relatively moist climate characterised by hot summers and mild to cool winters (Nanson 1986, MCMC 1996, BOM 2015). Annual rainfall for the local area (i.e., Wingham) is approximately 1100 mm and is mostly summer dominant, with the wettest months being January to April and the driest months experienced from July through to October (Table 3.1) (Hughes & Watkins 2011, BOM 2015). Summer rainfall events may vary in intensity and are typically short lasting (BOM 2015).

Table 3.1. Wingham monthly rainfall (mm) including years between 1888-2015.

Statistic	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Mean	125	139	145	99	81	86	59	53	56	78	86	100	1100
Lowest	7	0	2	2	4	0	0	0	0	0	4	7	492
Median	103	102	116	67	55	56	37	33	38	62	77	90	1079
Highest	693	813	716	456	372	529	271	333	252	413	346	391	2212

BOM station number: 060036 (latitude: 31.86°S, longitude: 152.34°E, elevation: 66 m).

The Manning River has a mean annual discharge of 1, 854 GL/yr (Hughes & Watkins 2011). Sources of water entering the Manning River from highest to lowest contributions,

respectively, include: catchment runoff, wetlands, and groundwater (Hughes & Watkins 2011). Groundwater storages within the catchment are largely confined to the narrow floodplain pockets, which may also contain small wetlands; for example: as back swamps or old channels (Hughes & Watkins 2011).

3.4. Study site: 'Dairy A'

It has been requested that the identity of the dairy farm chosen for this study remain anonymous. In order to respect this decision, the property under examination will be referred to as Dairy A for the purpose of this text. In addition, no names or explicit geographic location (i.e., geographic coordinates) will be included in the site description.

Dairy A is situated on a parcel of land confined by a large meander of the Manning River, upstream of Wingham (Figures 3.1, 3.2 and 3.3). A significant portion of the farm is comprised of pastured alluvial flats and adjoining slopes, both primarily utilised for grazing and milk production. The dairy is approximately 200 hectares (2 km^2) in size, operating with roughly 700 milking cows (i.e., 'milkers'). Thus, Dairy A may be considered an intensified operation in relation to many other dairies throughout Australia (ABARES 2015). As a result, local water catchment authorities (MidCoast Water) have prioritised this dairy amongst others in ensuring that the overall operation remains adequately sustainable. For example, through supplying funding needed to install an upgraded recycled waste system, enabling the dairy to recycle waste generated from the milking process, thereby, providing a more sustainable means of disposal. This includes both application of effluent through irrigation and dried solid waste via a spreader-type tractor implement. Since the implementation of recycled waste practices, however, there has been little follow up investigation into how effectively they are being utilised and whether or not nutrient contributions from Dairy A to the Manning River should remain a concern.

Dairy A is set within a landscape consisting of an assemblage of stepped alluvial units with varying height and (tentatively inferred) age above the existing channel (Figures 3.2 and 3.3). Hence, this site was well suited for a case study approach to be adopted, enabling key findings to potentially have both localised (i.e., specific to the farm under study) as well as broad scale implications, as was highlighted in Chapters 1 and 2.

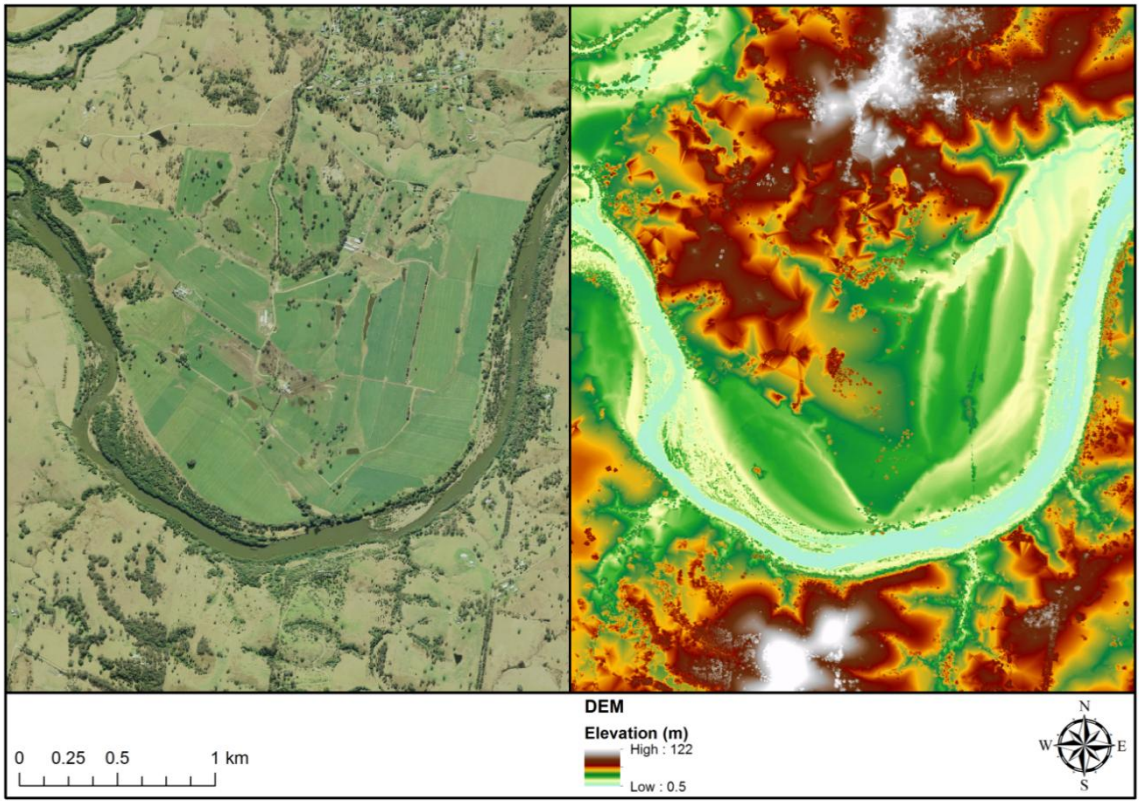

Figure 3.2. Dairy A: Airborne imagery (left) and Lidar-derived digital elevation model (DEM) (right).

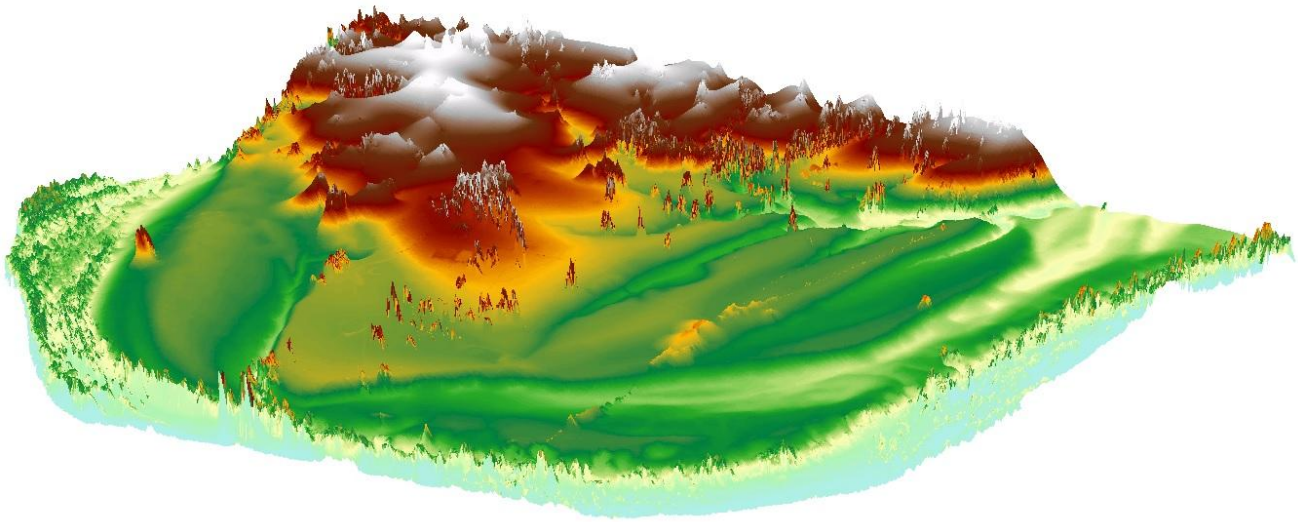

Figure 3.3. 3D model of Dairy A generated using DEM (Figure 3.2) in *ArcScene*.

3.5. Sample collection

3.5.1. Soils

Targeted sites for sampling included areas of the farm with varying soil characteristics and fertiliser use practices, such as, effluent versus chemical fertiliser treatments as well as control samples with no, or minimal (deliberate) phosphorus inputs. Due to a lack of suitable control soils at the study site, some control samples were taken from another property located across the river (Appendix XVI). Fertiliser use information was obtained prior to carrying out the field work through consultation with the farm manager(s), whereas variations in soil characteristics were predicted based on major variations in surface elevation (e.g., Figure 3.4) above the existing channel (i.e., Manning River).

Figure 3.4. Example of major surface elevation change at the study site, indicating potentially variable soil characteristics between the lower (foreground – T_1) and higher (middleground – T_2) surface.

A number of sampling locations of interest were predetermined before entering the field using geographical information system (GIS) software '*ArcGIS'*. Points were located in the field using a 'Trimble geo XH' global positioning system (GPS). However, not all locations were accessible (e.g., due to fencing layout, track conditions, livestock, etc.), therefore, actual sites sampled were determined on an *ad-hoc* basis, as a function of predefined locations of interest coupled with accessibility. All sampling locations were accurately recorded using the GPS in the field and later downloaded onto *ArcGIS*.

Sample collection for this study was carried out by means of coring, using a hand auger (1.5 m max depth) as well as a percussion corer, equipped with open as well as close core

barrels. Samples were taken at a range of depths (generally ranging from 0 – 100 cm). Cores extracted with the percussion corer were extracted within a 1 m grid to form two-core-composite samples: i.e., by mixing samples of matching depth ranges together.

One of the effluent cores was extracted using a closed barrel corer in order to preserve the natural soil profile stratigraphy as best as possible. This core was opened using the University of Wollongong (UOW) School of Earth and Environmental Sciences (SEES) recently constructed custom core cutter. Once opened, subsamples were taken at a relatively high resolution (e.g., every 5-10 cm) and then processed, prepared and analysed with the other samples.

3.5.2. Effluent

Effluent collection was carried out during both field trips (i.e., during March and June). Using a 3 m sampling pole equipped with a 1 L sampling vessel, effluent was collected from several different sections of the pond and combined together to form a composite sample by transferring into a separate (500 mL) vessel and mixing until homogenised. This process was repeated for each effluent sample collected.

3.6. Soil sample analysis

Soil sample analysis consisted of determining some parameters internally (i.e., using UOW facilities) whilst others were tested commercially. All commercial analyses were carried out by Southern Cross University Environmental Analysis Laboratory (EAL). Due to both time and funding constraints, not all samples were analysed commercially. Thus, groupings of representative samples were chosen to be analysed based on their fertiliser treatment history and location (i.e., modern floodplain, Terrace 1, etc.).

3.6.1. Sample Processing and Preparation

Upon arrival in the lab, representative 10-20 g (wet weight) subsamples were removed from each sample and double bagged to be used for the grain size analysis. The remainder of each sample was spread out onto clean plastic trays and broken up by hand into smaller clods. Organic material (e.g., roots and other plant residues) was removed by hand where practically possible. Samples were covered with newspaper to prevent dust contamination

and left to air-dry for approximately 1-2 weeks at room temperature (15-25 ^{o}C). Once dry, samples were manually crushed using a clean, dry ceramic mortar and pestle until small enough to pass through a 2 mm (20#) mesh sieve. The < 2 mm soil was then mixed by means of coning and quartering, and subsequently reduced to achieve a small, homogenised subsample (10-15 g). After bagging the remaining < 2 mm portion, the subsample was ground to pass through a 250 μm (60#) mesh sieve and then bagged and stored until required. Samples crushed to < 2 mm were used in all parameter analyses apart from mineralogy; for which the < 250 μm sample was used. Samples containing gravels, pebbles or rocks > 2 mm were not included in the analysis, as these types of materials are not typically considered actual 'soil' materials (vanLoon & Duffy 2010).

3.6.2. Laboratory methods

Moisture Factor

Various soil test results are often expressed on an oven-dry basis (105 ^{o}C) throughout the literature. For this study, however, the water extractable phosphorus (WEP) analysis was carried out using air-dried soils. In order to adjust the final results accordingly, it was therefore necessary to carry out a moisture factor analysis to determine the water content of air-dried samples. Through determining the air-dry moisture content (M%), air-dry soil test values were converted to oven-dry results using the equation below, derived from Rayment & Lyons (2011):

$$Oven-dry\ result = \frac{[Air-dry\ result\ \times (100 + M\%)]}{100}$$

Determination of the air-dry moisture content of soils was carried out in accordance with the procedure described in Rayment & Lyons (2011), section 2A1. This consisted of drying approximately 10 g (recorded to 4 decimal places) of air-dry sample (< 2 mm) in a soil moisture can at 105 ^{o}C for 24 hours, and then recording the change in weight after being cooled in a desiccator. The air-dry moisture content was calculated as follows:

$$M\% = \frac{Wt\ of\ moisture\ (g) x\ 100}{Wt\ of\ oven-dry\ soil\ (g)}$$

Grain size analysis

The grain size analysis was carried out using a Malvern Mastersizer 2000, at the University of Wollongong (UOW) School of Earth and Environmental Science (SEES) sedimentology laboratory, in accordance with the UOW standard operating procedure. This consisted of repeatedly introducing very small, representative portions of sample (< 0.5 g) into the 'mixing chamber' until there was enough evenly dispersed sample to obtain a sufficient beam obstruction level (indicated by the instrument software). Once this was achieved, grain size distributions were determined by the instrument via laser diffraction, where 5 measurements are made over a period of approximately 5 minutes to achieve an averaged result. To ensure sufficient quality control, the instrument flow cycle was rinsed thoroughly between each sample. In addition, the background was inspected prior to each sample being introduced and subsequently used to calibrate the instrument for each analysis.

Mineralogy

Mineralogy was determined via X-ray diffraction (XRD) analysis in the UOW SEES laboratories, following the standard UOW operating procedure. The analysis was carried out using a Phillips PW 1050/80 Goniometer with a Cu X-ray tube, powered by a Spellman DF3 generator running at 35 kilovolts and 28.8 milliamps to achieve an output of 1 kilowatt. Samples (< 250 µm) were prepared by crushing with an agate mortar and pestle to achieve a powder as fine as possible. Samples were analysed using the following instrument settings: range = 4^o to 70^o at 2^o per minute with a step size of 0.02^o. XRD traces were generated using a GBC 122 control system. Data was exported from the instrument software and processed using two analytical software packages: first, *Traces* was used to standardise the quartz peak of each sample, followed by *SIROQUANT* to quantify the mineral composition. Given the limited accuracy (i.e., semi-quantitative) of this method, mineralogy results should be interpreted with caution.

pH

Soil pH was determined commercially by EAL using an in-house method, based on Rayment & Lyons (2011) 4A1: "pH of 1:5 soil/water suspension". First, soils were mixed with distilled water at a 1:5 soil solution ratio by shaking for 1 h at 25 ^{o}C. Following this,

the final solution pH was measured (whilst continually being stirred) using a calibrated pH meter.

Electrical Conductivity

Electrical conductivity (EC) was determined commercially by EAL following the method described in Rayment & Lyons (2011) 3A1: "Electrical conductivity (EC) of 1:5 soil/water extract". Soils were mixed with distilled water at a 1:5 soil solution ratio at approximately 25 ^{o}C for 1 h to dissolve soluble salts. The soil solution was then allowed to settle for a minimum period of 30 minutes and EC of the settled supernatant was measured using a calibrated conductivity cell and meter.

Organic Carbon

Organic carbon (OC) was determined commercially by EAL in accordance with the Walkley-Black method, as per Rayment & Lyons (2011) 6A1:"Organic C – W&B". In summary, concentrated sulfuric acid (H_2SO_4) was added to soils (air-dry; < 0.5 mm) which had been previously wetted with 0.5 M sodium dichromate ($Na_2Cr_2O_7$) solution. Next, reagent water (refer to method) was added to effectively increase the temperature high enough (110-120 ^{o}C) to induce substantial oxidation, generating the following reaction:

$$2Cr_2O_7^{2-} + 16H^+ + 3C \rightarrow 4Cr^{3+} + 8H_2O + 3CO_2$$

Cromic ions (Cr^{3+}) produced from the reaction are proportional to the amount of OC oxidised; thus, allowing OC to be determined by measuring the absorbance of Cr^{3+} at 600 nm. The weight of soil used for the analysis was determined based on the expected C content of the sample, for example: 1.00 g for samples approximated to have < 5 % C, and 0.20 g for > 5 % C.

Oxalate extractable Iron and Aluminium

Oxalate extractable Fe (Fe_{ox}) and Al (Al_{ox}) concentrations were determined commercially by EAL, in accordance with Rayment & Lyons (2011) 13A1: "Oxalate-extractable iron, aluminium and silicon" (Rayment & Lyons 2011). In the dark, 1.0 g of air-dry soil was shaken with 100 mL of acid oxalate reagent for 4 h at 25 ^{o}C, and then centrifuged to obtain a clear supernatant. Extracts were diluted to achieve an appropriate dilution factor

and then analysed via inductively coupled plasma mass spectrometry (ICP-MS) following the EAL standard operating procedure.

Water Extractable Phosphorus

Water extractable phosphorus (WEP) was determined following a slightly modified version of the method described in Self-Davis *et al.* (2000): "Determination of Water- and/or Dilute Salt-Extractable Phosphorus". The procedure consisted of adding deionised water to soil at a 1:10 soil-solution ratio and shaking (end over end) for a period of one hour, followed by centrifuging samples at 7500 rpm for a period of 10 minutes. The supernatant obtained was then filtered into 15 mL plastic sampling tubes using a 25 mL syringe and 45 µm disposable filters.

Orthophosphate concentration of the supernatant was determined via flow injection analysis (FIA), using a LACHAT QuikChem 8500 FIA, based on a method of Murphey and Riley (1962) adapted for FIA. The final reagent concentrations used in the reaction were 0.6 $g.L^{-1}$ ammonium molybdate, 1 $g.L^{-1}$ ascorbic acid, 0.05 $g.L^{-1}$ potassium antimony tartrate and 0.125M H_2SO_4, with a reaction time of approximately 80 seconds. The absorbance of the phosphorus-molybdenum-antimony complex was measured at 880 nm, yielding a concentration value for molybdenum reactive P based on the calibration standards.

Prior to use, all equipment used in the analysis was washed thoroughly with tap water followed by copious amounts of deionised water; this cleaning procedure was deemed sufficient enough to negate contamination concerns. For quality assurance purposes, each run of samples (generally n = 20) consisted of at least one method blank, one run blank and several sample replicates. Raw concentration results were blank corrected and converted from air-dry to oven-dry result using the 'moisture factor' following the method described above.

Total Phosphorus

Total phosphorus was determined commercially by EAL following the method of Rayment & Lyons (2011) 17C1: "Pseudo-total elements of soils and sediments – conventional aqua regia block digestion and determination by atomic spectroscopy". In summary, soils underwent a temperature-controlled digestion using a concentrated 'aqua

regia' (1:3 HNO_3 to HCl) acid solution to obtain a digestate which was decanted, diluted and analysed via inductively coupled plasma atomic emission spectroscopy (ICP-AES), following the EAL standard operating procedure.

Colwell Phosphorus

Colwell phosphorus was determined commercially by EAL, following the method of Rayment & Lyons (2011) 9B2: "Bicarbonate-extractable P (Colwell-P) – automated colour, continuous segmented flow/FIA". Briefly, 1.00 g air-dry soil (< 2 mm) was mixed end-over-end with 100 mL extracting solution (0.5 M $NaHCO_3$ at pH 8.5) at 25^{o}C for a period of 16 h and then centrifuged (or filtered). Orthophosphate was then determined via FIA, carried out in accordance with the standard EAL operating procedure.

Phosphorus Buffering Index (PBI)

PBI was determined commercially following the method of Rayment & Lyons (2011) 9I2b: "P Buffer index – PBI_{+ColP} – ICP-AES". To 7.0 g of air-dry soil (< 2 mm), 70 mL of P equilibrating solution (containing 100mg $P.L^{-1}$) and 3 drops of chloroform ($CHCl_3$) were added. Samples were then shaken continuously end-over-end for a period of 17 h at 25^{o}C, followed by centrifuging at 3000 rpm (and/or filtering) in order to obtain a particulate free supernatant. Solutions were analysed via ICP-AES to determine the total P concentration, which was then used to calculate the amount of 'freshly sorbed P' by subtracting the total P measured in solution from the original concentration of P added to the soil. The PBI_{+ColP} was calculated using the following equation, taken from Rayment & Lyons (2011):

$$PBI_{+ColP} = \frac{[freshly\ sorbed\ P\ (mg\ P.kg^{-1})\ +\ Colwell\ P\ (mg.kg^{-1})]}{Final\ solution\ P\ conentration\ (mg\ P.L^{-1})^{0.41}}$$

3.7. Effluent sample analysis

Effluent analysis was conducted commercially by EAL. Only samples taken from the pond which is irrigated from (i.e., pond 2) were chosen for analysis. This included two samples, each with noticeably different total suspended solids contents. These particular samples were selected in order to obtain results more representative of the natural variability of effluent within the pond.

3.7.1. Laboratory Methods

pH

Effluent pH was determined commercially by EAL following the method described in APHA (2005) 4500-H^+-B: "Electrometric Method". Using a calibrated pH meter, pH was measured by placing the electrode in the sample and gently stirring until equilibrium was achieved, in order to obtain sample homogeneity and, in turn, a stable pH value result.

Salinity

Effluent salinity was determined commercially by EAL in accordance with APHA (2005) 2520B: "Electrical Conductivity Method". Using a calibrated salinity/conductivity meter, salinity was measure by placing the probe in the effluent solution and gently stirring to achieve a stable result.

Total Phosphorus

Total phosphorus was determined commercially by EAL in accordance with APHA (2005) 4500 P-H: "Manual Digestion and Flow Injection Analysis for Total Phosphorus". Samples were digested in a concentrated acid solution, diluted, and analysed via FIA following the EAL standard operating procedure.

Orthophosphate

Orthophosphate (PO_4^{3-}) was determined commercially by EAL as per APHA (2005) 4500 – P-G: "Flow Injection Analysis for Orthophosphate". Samples were centrifuged and/or filtered to remove suspended solids and then analysed via FIA following the EAL standard operating procedure.

3.8. Data processing and statistical analysis

All data obtained was summarised in a table divided by core locations with all parameters included. The only data requiring post-processing were the WEP results; where, the original result was corrected using the moisture factor (as above) so that it could be expressed on an oven-dry basis.

Data for individual cores were divided into groups based on the determined alluvial unit within which they were located, these included: (a) Modern floodplain (MF), (b) Terrace 1 (T_1), (c) Terrace 2 (T_2), and (d) Terrace 3 (T_3). In addition, the T_2 samples were further divided into control and effluent treated soils for various analyses. For most of the data analysis, soil data for each major alluvial unit was divided into three depth ranges: (1) 0-10 cm (surface), (2) 0-50 cm (subsurface), and (3) > 50 cm (deep), although, whole profile data was used for some of the statistical analyses. Mean values determined for various soil properties within the above groups were used to construct histograms to illustrate findings in a way which allowed both location and depth range of soils to be visually compared.

All statistical analyses were conducted using *IBM SPSS Statistical Software package version 20 for Windows*. Firstly, all data groupings were checked for normality using the 'Shapiro-Wilk' test, where the null hypothesis assumes the data is normal. Due to the limited sample size, most groupings of data were significantly different from a normal distribution ($p > 0.05$); thus, the null hypothesis was rejected. Although some data groups were found to have normal distributions, only non-parametric testing methods were used for the purpose of this study in order for the statistical methodology and associated results to remain consistent.

Two non-parametric tests were used in order to assess both variations and correlations between soil properties, location (i.e., between major alluvial units), and depth within the soil profile. The 'Kruskal-Wallis' test - considered the non-parametric equivalent to the 'analysis of variance' (ANOVA) testing method – was used in order to compare means between groups. The 'Spearman Rank-Order Correlation' test was used to reveal correlations between soil properties and location as well as depth.

Given the limitations of the project, the number of samples analysed were not large enough to carry out any (independently) reliable statistical analysis. As a result, only a limited statistical analysis has been provided in attempt to support the inferences made through non-statistical based methods; i.e., graphing. Thus, statistical outputs used to support the results should be interpreted with caution. In addition, it is worth noting that several total P result values were below the limit of detection (Appendix I); these values were reported as the detection limit (i.e., 50 mg/kg) in order to make it possible to include these samples in the various data analyses.

CHAPTER 4

Results

Chapter 4 presents the overall results of the study. The results obtained fall into several main categories, and have consequently been divided into four major sections. Section one (4.1) contains results indicating the variations in key soil properties between the four major alluvial units (MF, T_1, T_2 and T_3) under investigation. Section two (4.2) includes results pertaining to the influence of variable alluvial soil profiles on phosphorus (P) dynamics. Section three (4.3) presents results used to indicate the implications of variable soil profiles on P management. Section four (4.4) provides results used to assess the sustainability of Dairy A and potential risk of farm-P losses to the environment.

4.1. Variations in alluvial soil properties between major alluvial units

Within the farm boundary, several major terraces with variable heights above the existing channel were seen to occur ($T_1 – T_3$), with only one modern floodplain (MF) unit situated on the western bank. Boundaries for the major alluvial units and sites sampled are shown in Figure 4.1.

Soil analysis results revealed that there were a number of variations found to exist between soil profiles derived from each of the major alluvial units investigated (i.e., MF - T_3), including: grain size, mineralogy and oxalate extractable iron (Fe_{ox}) and aluminium (Al_{ox}). Variations between other tested parameters such as pH, EC and OC were assumed to be disproportionately influenced by landuse practices, thereby, rendering them of minimal interest for the purpose of this section (4.1).

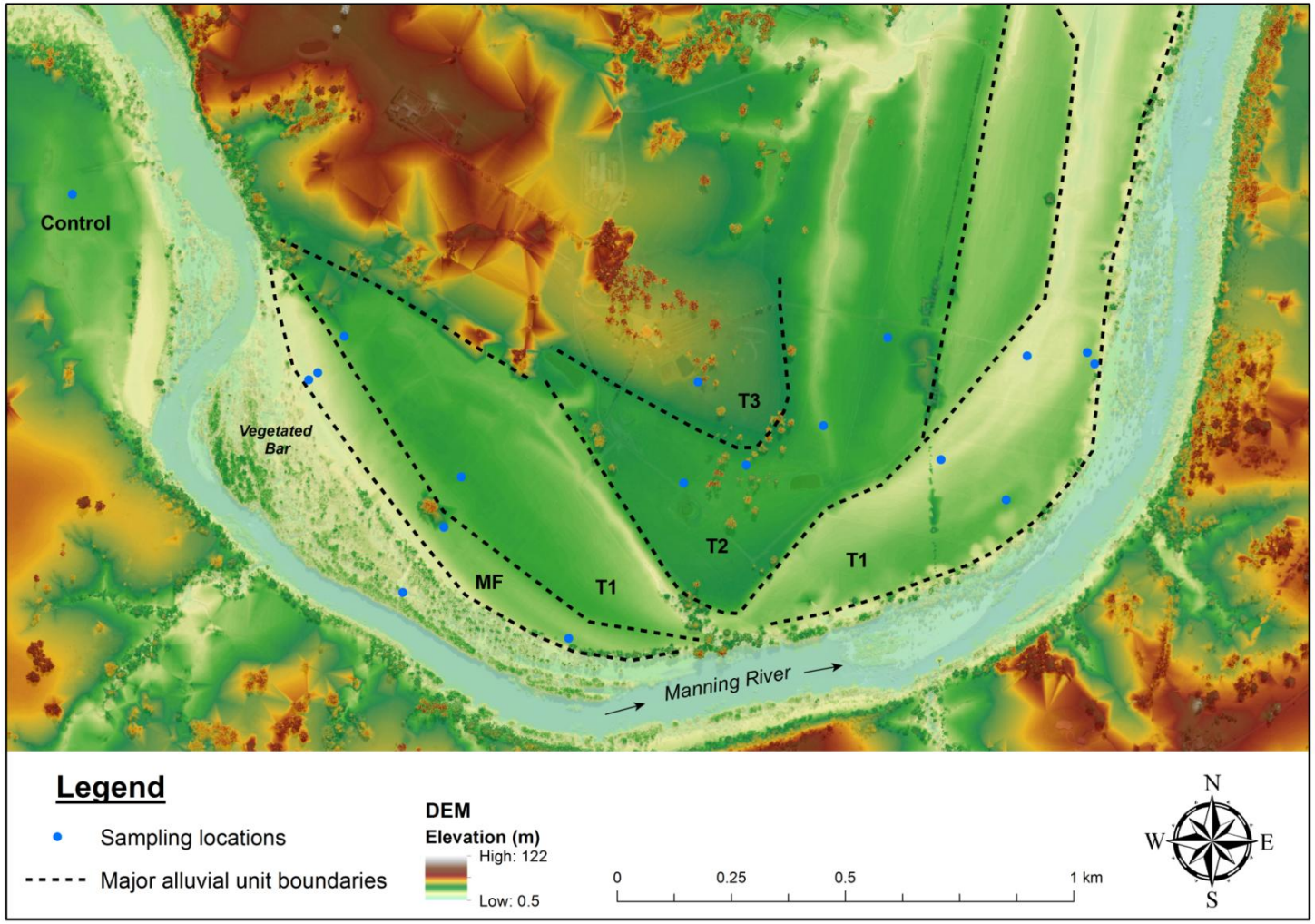

Figure 4.1. Image of the study site (Dairy A) overlain by a digital elevation model (DEM) derived from LiDAR data. Flow is from left to right.

4.1.1. Grain size

A summary of the calculated average grain size analysis results determined for the four major alluvial units of variable heights (Figure 4.1) are shown in Table 4.1.

Table 4.1. Summary of mean grain size analysis results determined for individual alluvial units

Alluvial Unit	Depth Range	% Sand	% Silt	% Clay	% Clay (2 µm)	*D [4, 3] (µm)	N
MF	Surface	43	51	7	1	120	4
	Subsurface	45	46	10	2	121	8
	Deep	62	27	11	5	221	12
T_1	Surface	29	63	9	1	76	6
	Subsurface	39	49	12	2	105	10
	Deep	46	32	22	15	131	11
T_2	Surface	27	64	9	2	70	5
	Subsurface	21	62	18	8	56	12
	Deep	15	53	32	22	45	19
T_3	Surface	28	63	9	2	76	1
	Subsurface	27	46	27	14	75	4
	Deep	32	44	24	15	107	7

Terms used to group depth fractions are representative of the following depth ranges: surface = 0-10 cm; subsurface = 10-50 cm; and, deep = > 50 cm.
*D [4, 3] represents volume weighted mean grain size; N = number of samples used to calculate average

Statistical analysis of the data revealed that the (< 2 µm) clay content in soils demonstrated a modest correlation with location (r_s = 0.489, p = < 0.001) (Appendix V). Through additional interpretation of the results presented in Table 4.1 and Figure 4.2, it was discovered that the (< 2 µm) clay content present in soils progressively increased from the MF through to the T_2 alluvial unit. Successive increases were found to be significant ($p < 0.05$) among the aforementioned suite of alluvial units, but, the T_3 soils were found to have similar (< 2 µm) clay contents to both T_1 and T_2 soils at various depths, (cumulatively) resulting in a p value > 0.05 (Appendix III). Sequential increases in (< 2 µm) clay content between the four major alluvial units were more prominent within the subsurface (10-50 cm) and deep (> 50 cm) depth ranges of soil profiles as compared to the surface soils (0-10 cm), which showed no significant variation among all four major alluvial units (p = > 0.05) (Table 4.1, Figure 4.2, Appendix III). Hence, significant differences in (< 2 µm) clay were only found in the subsurface and deep depth ranges of the soil profiles, whereas surface clay content was found to be relatively constant among all of the major alluvial units (Table 4.1, Figure 4.2). Further statistical analysis of the data

revealed that clay content had a positive correlation with depth in most of the major alluvial units ($p < 0.05$), excluding T_3 ($p = 0.727$) (Appendix VI).

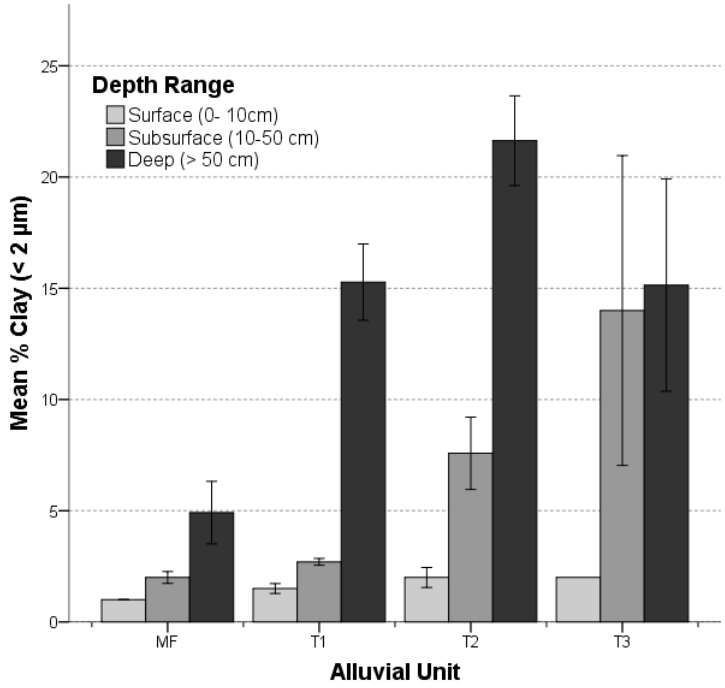

Figure 4.2. Histogram illustrating mean % clay (< 2 µm) content in surface, subsurface and deep soils sampled from the four major alluvial units (Figure 4.1). Error bars indicate ±1 standard error of the mean (refer to Table 4.1 for additional information regarding the input data).

Sand content was found to differ between the four major alluvial units (Figure 4.3; Appendix III) in a trend indicating a sequential decrease in sand from MF through to T_3 soils ($r_s = -668$, $p = < 0.001$) (Appendix V). The most variance in sand content between alluvial units was found within the deep range of the soil profile, followed by the subsurface, and then surface soils, which showed no significant variation ($p > 0.05$) (Appendix III). Sand content was correlated with soil depth for MF and T_2 soils ($p = 0.007$ and 0.012, respectively); where, MF soil profiles displayed a positive trend ($r_s = 0.640$, $p = 0.001$) suggesting an increase in sand with depth, whereas T_2 soil profiles displayed a negative trend ($r_s = -0.500$, $p = 0.002$) indicating a general decrease in sand with depth (Appendices IV and VI). No significant correlation between sand and depth was found for the T_1 and T_2 soil profiles ($p > 0.05$) (Appendix VI). However, despite the statistical evidence, Figure 4.3 suggests that a positive trend between sand and depth may be apparent for the T_1 soil profiles as well.

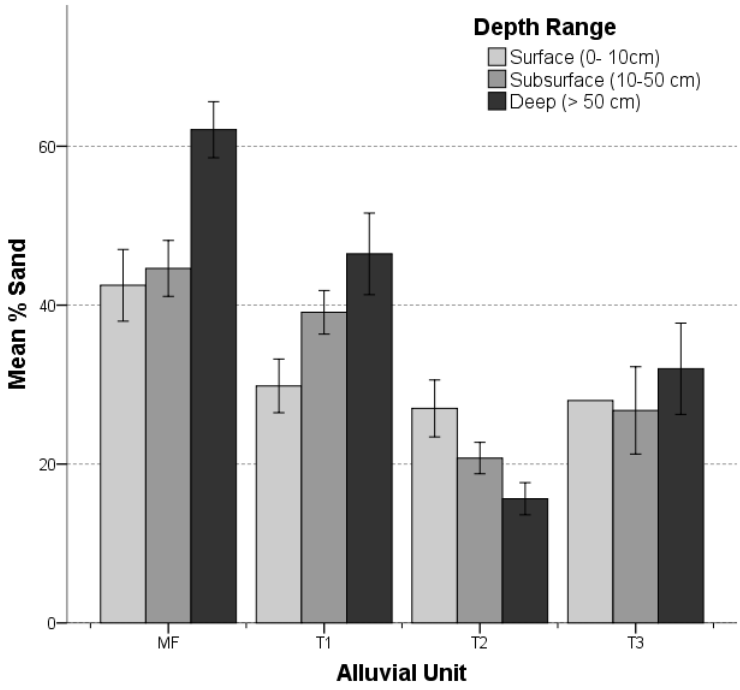

Figure 4.3. Histogram illustrating mean % sand content in surface, subsurface and deep soils sampled from the four major alluvial units (Figure 4.1). Error bars indicate ±1 standard error of the mean (refer to Table 4.1 for additional information regarding the input data).

Mean grain size, defined as the volume weighted mean D [4, 3], was found to correlate negatively with location (r_s = -0.571, p = < 0.001). Through visual interpretation of Figure 4.4, it is evident that this correlation represents a successive decrease in mean grain size from MF soils through to T_2 and T_3 soils. Mean grain size increased with depth for MF soils (r_s = 0.598, p = 0.002), decreased with depth for T_2 soils (r_s = -0.473, p = 0.004) and had no significant relationship with depth (p = > 0.05) for T_1 and T_3 soil profiles (Appendix VI).

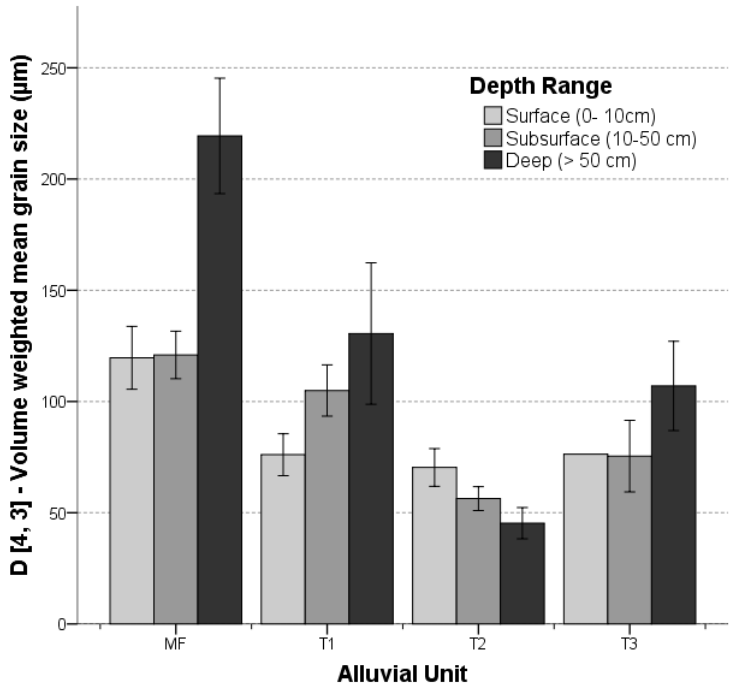

Figure 4.4. Histogram illustrating D [4, 3] volume weighted mean grain size (µm) in surface, subsurface and deep soils sampled from the four major alluvial units (Figure 4.1). Error bars indicate ±1 standard error of the mean (refer to Table 4.1 for additional information regarding the input data).

The degree of particle size differentiation (i.e., ratio of B/A horizon clay content; Appendix VII) tended to gradually increase from the MF through to T_2 soil profiles, but, no apparent change was seen when comparing T_2 and T_3 soil profiles (Figure 4.5). Conversely, statistical methods indicated there was no significant correlation between particle size differentiation and location (i.e., between the four major alluvial units) ($p = 0.084$); however, this result was significant at the 0.1 level (Appendix V). Similarly, statistical analysis of the results also suggested that there was no significant difference in the particle size differentiation between major alluvial units ($p > 0\ .05$), which might also be inferred from the standard error bars shown in Figure 4.5 (Appendix III).

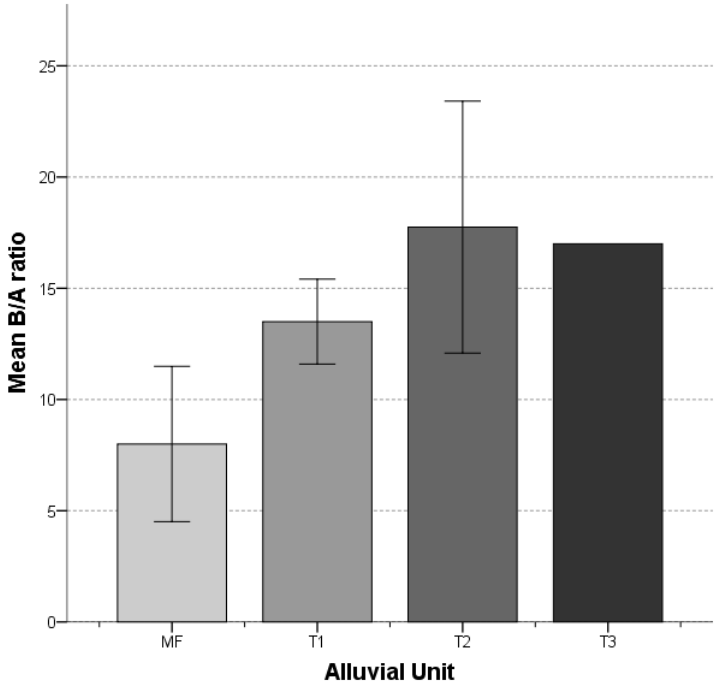

Figure 4.5. Histogram illustrating the average degree of particle size differentiation for each of the four major alluvial units (Figure 4.1), indicated by the B/A ratio calculated using the (< 2 µm) clay grain size data for all soils (Appendix VII). Error bars represent ±1 standard error of the mean.

4.1.2. Mineralogy

Quartz content was found to be moderate to high for all samples, with the vast majority of samples falling into the high (i.e., > 40%) category (Figure 4.6; Appendix II). Clay minerals illite and kaolin were seen to become progressively more abundant from the MF soils through to the T_3 soils (Figure 4.6; Appendix II). Additionally, goethite and gibbsite followed the same trend (Figure 4.6).

The relative abundances of illite and kaolin minerals were also found to vary between the four major alluvial units in a pattern where the general abundance of kaolin tended to become increasingly more dominant from the T_1 through to the T_3 soils (Figure 4.6).

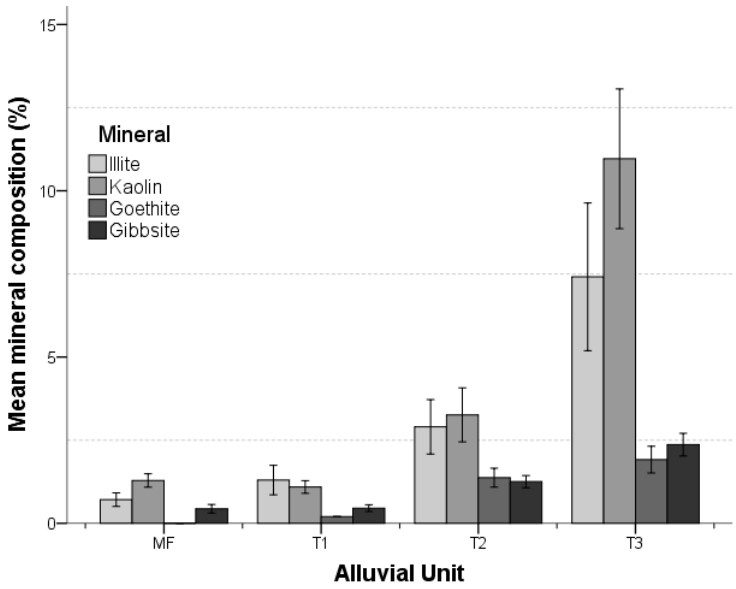

Figure 4.6. Histogram illustrating mean % mineral composition of the four major alluvial units (Figure 4.1), derived from a range of representative samples. Error bars represent ±1 standard error of the mean (refer to Appendix II for additional information regarding the input data).

4.1.3. Oxalate extractable iron and aluminium

Oxalate extractable aluminium (Al_{ox}) concentrations were found to increase from the MF through to the T_3 soils, but, only for the subsurface and deep depth ranges of the soil profile (Figure 4.7, Appendix I). Complimentary statistical analysis supported this by revealing a positive correlation between Al_{ox} and location (r_s = 0.562, p = < 0.001, Appendix V). Statistical methods also showed that the cumulative mean Al_{ox} concentration for all depth ranges was found to be significantly lower in the MF soils compared to the three terrace units ($p < 0.05$), whereas no significant difference was found between T_1, T_2 and T_3 soils ($p > 0.05$) (Appendix III). No consistent results could be identified which indicated a correlation between Al_{ox} and depth among all four major alluvial units (Appendix VI). However, the mean Al_{ox} determined for each major alluvial unit was generally higher in either the subsurface or deep depth range of the soil profile (or both) as compared to the surface soil concentration for all locations (Figure 4.7; Appendix I).

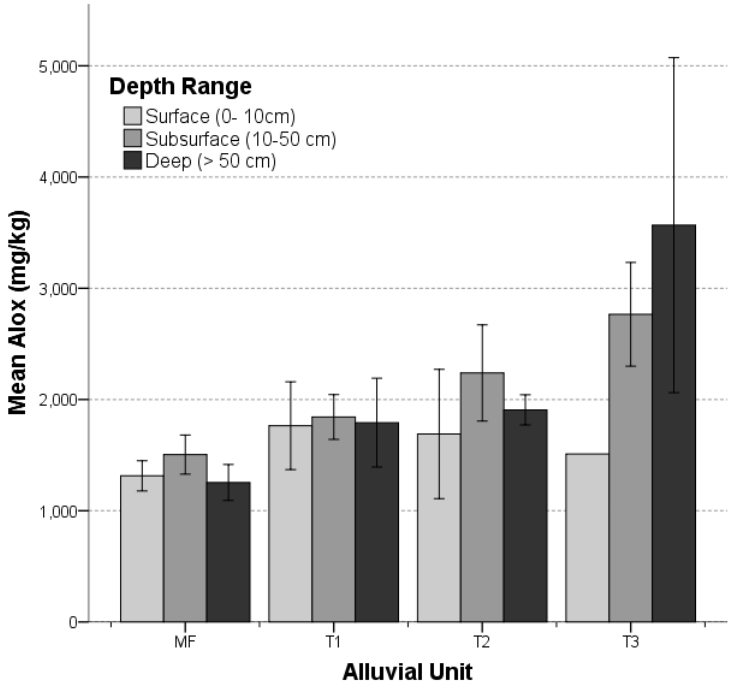

Figure 4.7. Histogram illustrating the mean oxalate extractable aluminium (Al_{ox}) calculated for surface, subsurface and deep soils sampled from the four major alluvial units (Figure 4.1). Error bars indicate ±1 standard error of the mean (refer to Appendix IX for additional information regarding the input data).

Statistical methods revealed that oxalate extractable iron (Fe_{ox}) was moderately correlated with location (r_s = -515, p = 0.001; Appendix V), but, based on a visual interpretation of Figure 4.8, it is unlikely that this relationship reflects the same pattern revealed for Al_{ox}. It was discovered that there was a strong negative correlation between Fe_{ox} and depth for the T_3 soil profile (r_s = -0.905, p = < 0.001). Figure 4.8 shows similar trends of decreasing Fe_{ox} with depth for the other three alluvial units (MF, T_1 & T_2), but, for these soils no statistically significant relationship was observed (Appendix VI).

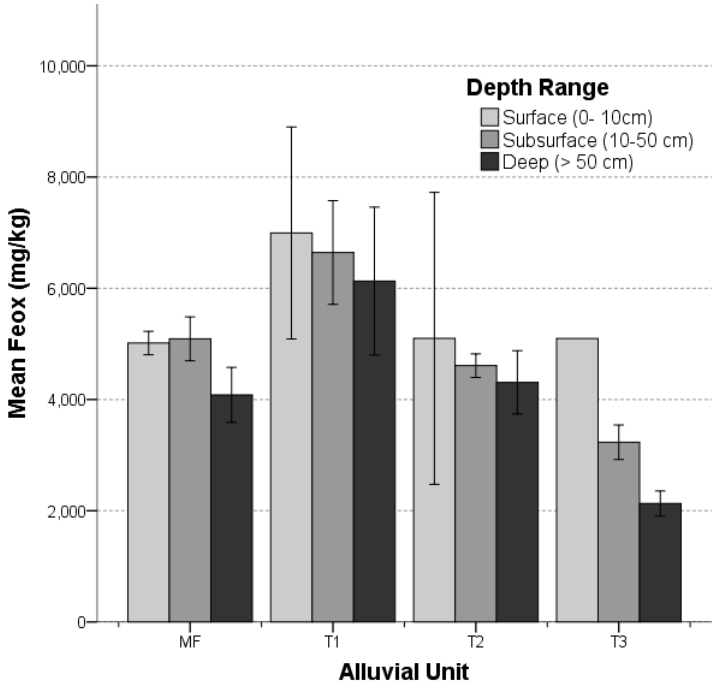

Figure 4.8. Histogram illustrating the mean oxalate extractable iron (Fe_{ox}) concentration calculated for surface, subsurface and deep soils sampled from the four major alluvial units (Figure 4.1). Error bars indicate ±1 standard error of the mean (refer to Appendix IX for additional information regarding the input data).

4.1.4. Summary

A number of variations in key soil properties between the four major alluvial units were revealed. From these variations, a range of important patterns reflecting progressive changes in soil properties from the MF through to the T_3 soils were able to be identified. These included:

(1) Increases in (< 2 µm) clay content

(2) Decreases in sand content and mean grain size

(3) Transition from upward-fining to downward-fining stratification of alluvium

(4) Increase in the degree of particle size differentiation (i.e., B/A ratio)

(5) Increases in kaolin and illite, and to a lesser extent goethite and gibbsite

(6) General increase in the dominance of kaolin over illite

(7) Increasing Al_{ox} concentration

Variations were found to be most prominent within the subsurface (10-50 cm) and deep (> 50 cm) depth ranges of the soil profiles, whereas surface soils showed minimal variation across all alluvial units; hence, surface soils (0-10 cm) do not necessarily represent the patterns of change in key soil properties expected to influence P dynamics.

4.2. Influence of soil profile variations on phosphorus dynamics

Phosphorus buffering index (PBI) values determined for all commercially analysed samples (excluding control samples) demonstrated a positive correlation with (< 2 µm) clay content (r_s = 0.680, p = < 0.01) and Al_{ox} (r_s = 0.822, p = < 0.001) (Table 4.2). Conversely, PBI was negatively correlated with total phosphorus (r_s = -0.577, p = < 0.010), Colwell phosphorus (r_s = -0.417, p = < 0.007), sand content (r_s = -0.831, p = < 0.001), mean grain size (r_s = -0.698, p = < 0.001) and pH (r_s = -0.599, p = < 0.001) (Table 4.2). Statistical analysis of data obtained from the control samples revealed largely similar correlations to those stated above (Table 4.3), but, several correlations were found to be inconsistent, including, pH (r_s = -0.426, p = 0.116), Colwell phosphorus (r_s = -0.447, p = 0.095) and sand content (r_s = -0.467, p = 0.079).

In addition to these findings, Tables 4.2 contains a number of other key significant correlations, including: WEP and EC (r_s = 0.580, p = < 0.001), WEP and Colwell P (r_s = 0.394, p = 0.014), and PBI and OC (r_s -0.340, p = 0.030). These results, however, will be further discussed in Chapter 6.

Table 4.2. Statistical analysis results indicating correlations between soil properties and phosphorus related parameters for all commercially analysed samples determined via the Spearmans Rank-Order Correlation test (see bottom of Table 4.3 for additional information on table components).

		PBI	TP	CP	WEP	Sand	Clay	Silt	D [4, 3]	Al_{ox}	Fe_{ox}	pH	OC	EC
PBI	r_s	1	-0.577	-0.417	0.213	-0.831	0.680	0.461	-0.698	0.822	-0.240	-0.599	-0.340	0.599
n = 40	sig.	.	0.000*	0.007*	0.188	0.000	0.000*	0.003*	0.000*	0.000*	0.140	0.000*	0.030*	0.000*
TP	r_s	-0.577	1	0.837	0.156	0.550	-0.734	0.014	0.416	-0.389	0.579	0.333	0.830	-0.213
n = 40	sig.	0.000*	.	0.000*	0.348	0.000*	0.000*	0.929	0.008*	0.013*	0.000*	0.036*	0.000*	0.186
CP	r_s	-0.417	0.837	1	0.394	0.323	-0.491	0.088	0.210	-0.304	0.450	0.437	0.813	-0.004
n = 40	sig.	0.007*	0.000*	.	0.014*	0.042*	0.001*	0.591	0.194	0.057	0.004*	0.005*	0.000*	0.979
WEP	r_s	0.213	0.156	0.394	1	-0.128	-0.012	0.131	-0.060	0.001	-0.180	-0.121	0.188	0.580
n = 38	sig.	0.188	0.348	0.014*	.	0.445	0.944	0.434	0.719	0.996	0.283	0.468	0.258	0.000*

Table 4.3. Statistical analysis results indicating correlations between soil properties and phosphorus related parameters for all control samples determined using the Spearmans Rank-Order Correlation test.

		PBI	TP	CP	WEP	Sand	Clay	Silt	D [4, 3]	Al_{ox}	Fe_{ox}	pH	OC	EC
PBI	r_s	1	-0.540	-0.447	-0.005	-0.467	0.567	-0.225	-0.644	0.913	-0.050	-0.426	-0.190	0.097
n = 5	sig.	.	0.038*	0.095	0.985	0.079	0.027*	0.421	0.010*	0.000*	0.859	0.116	0.499	0.730
TP	r_s	-0.540	1	0.631	-0.082	0.620	-0.852	0.452	0.811	-0.547	0.490	0.157	0.670	0.03
n = 5	sig.	0.038*	.	0.012*	0.771	0.014*	0.000*	0.091	0.000*	0.035*	0.064	0.577	0.006*	0.914
CP	r_s	-0.447	-0.447	1	0.124	0.075	-0.422	0.730	0.341	-0.488	0.486	0.532	0.826	0.341
n = 5	sig.	0.095	0.095	.	0.659	0.790	0.117	0.002*	0.213	0.065	0.066	0.041*	0.000*	0.214
WEP	r_s	-0.005	-0.082	0.124	1	0.025	-0.032	0.134	-0.082	-0.252	-0.160	0.165	-0.070	0.486
n = 5	sig.	0.985	0.771	0.659	.	0.929	0.909	0.634	0.770	0.365	0.580	0.557	0.800	0.066

TP = total P, CP = Colwell P, WEP = water extractable P; PBI = phosphorus buffering index; OC = organic carbon; EC = electrical conductivity; D [4, 3] = volume weighted mean grain size

r_s = Spearmans rank correlation coefficient; * denotes correlation is significant at the 0.05 level (2-tailed); n = number of samples

The cumulative mean PBI concentrations for all depth ranges in soil profiles were observed to differentiate between the four major alluvial units, following a pattern whereby the cumulative mean PBI increased sequentially from MF through to T_3 soils (Figure 4.9).

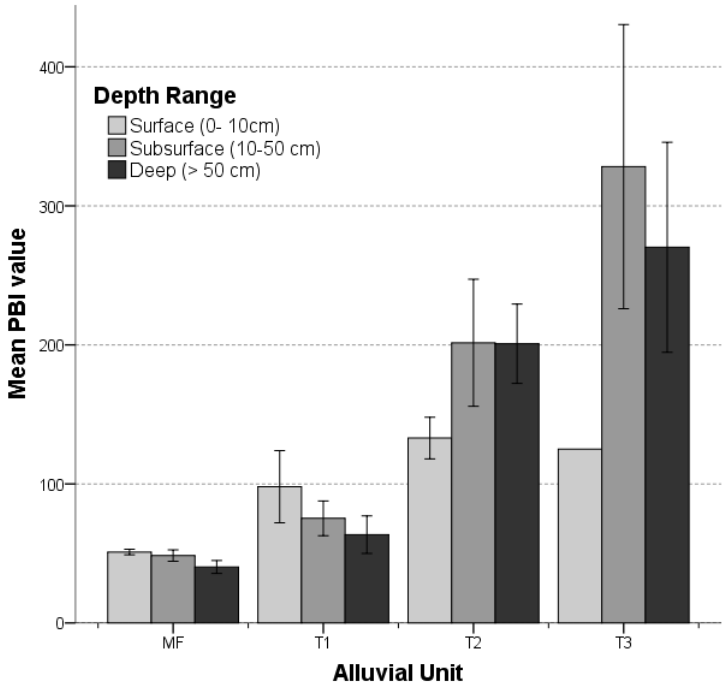

Figure 4.9. Histogram illustrating the mean phosphorus buffering index (PBI) calculated for surface, subsurface and deep soils sampled from the four major alluvial units (Figure 4.1). Error bars represent ±1 standard error value from the mean (refer to Appendix XI for additional information regarding the input data).

Statistical methods revealed that there was a significant difference in PBI between all major alluvial units within the subsurface and deep depth ranges soil profiles ($p = 0.011$, effect size = 74 % and $p = 0.010$, effect size = 71%, respectively), but not between surface soils ($p > 0.05$). Additional *post-hoc* statistical analysis revealed that the cumulative mean PBI for the T_2 and T_3 soils were not statistically different from one another at the 0.05 significance level ($p = 0.391$).

Therefore, through visual interpretation of Figure 4.9, in conjunction with results obtained from the statistical analysis of the data, it was discovered that the PBI varied between the four major alluvial units of varying height (Figure 4.1) in a pattern reflecting the following trend:

$$\text{PBI: MF} < T_1 < T_2 \leq T_3$$

In addition, it was also found that PBI values were increasingly more variable with depth from the MF through to the T_3 soils (Figure 4.10). Thus, the pattern of PBI with depth between major alluvial units was found to be consisted with particle size differentiation results (Figure 4.5).

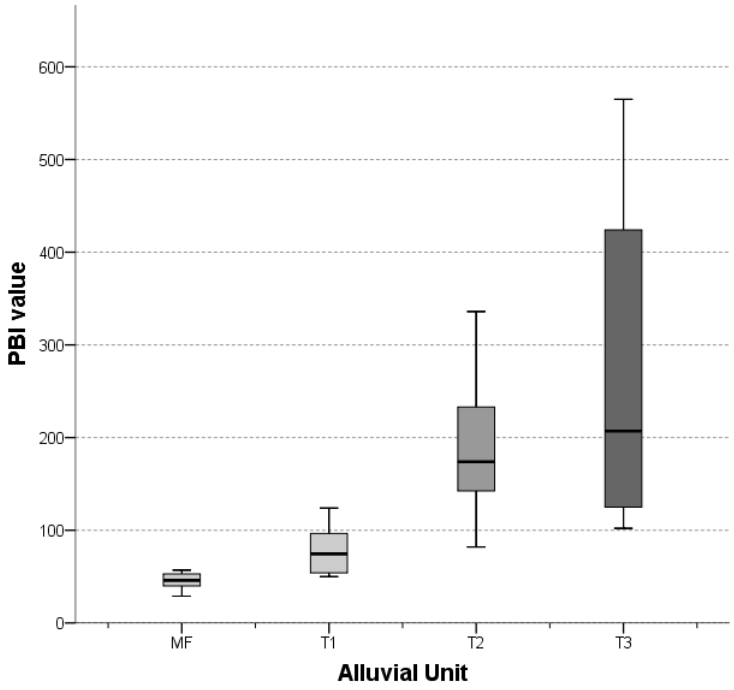

Figure 4.10. Box plots indicating the differentiating variation of determined PBI values between the four major alluvial units (Figure 4.1). Refer to Appendix I for additional information regarding the input data.

4.2.1. Summary

PBI was found to be correlated with most of the key soil properties (including control samples) included in section 4.1., and progressively increased from the MF soils through to the T_3 soils (although T_2 and T_3 were not statistically different). This pattern, however, was found to be largely attributable to results derived from soils taken from the subsurface and deep depth ranges of soil profiles. In addition, the range of PBI values tended to increase from the MF through to the T_3 soils, indicating a progression in the extent of soil profile variation.

4.3. Implications for phosphorus management

Critical Colwell P values for each of the four major alluvial units (including all three depth ranges) were determined using the mean PBI values shown in Figure 4.9 (Figure 4.11, Appendix XIII). Figure 4.11 suggests that inputs of phosphorus required to achieve maximum production levels are different between all of the major alluvial units. The order of which relative levels of phosphorus inputs required for maximum production may be expected to occur is shown in the following sequence:

$$\text{P required: } MF < T_1 < T_2 \leq T_3$$

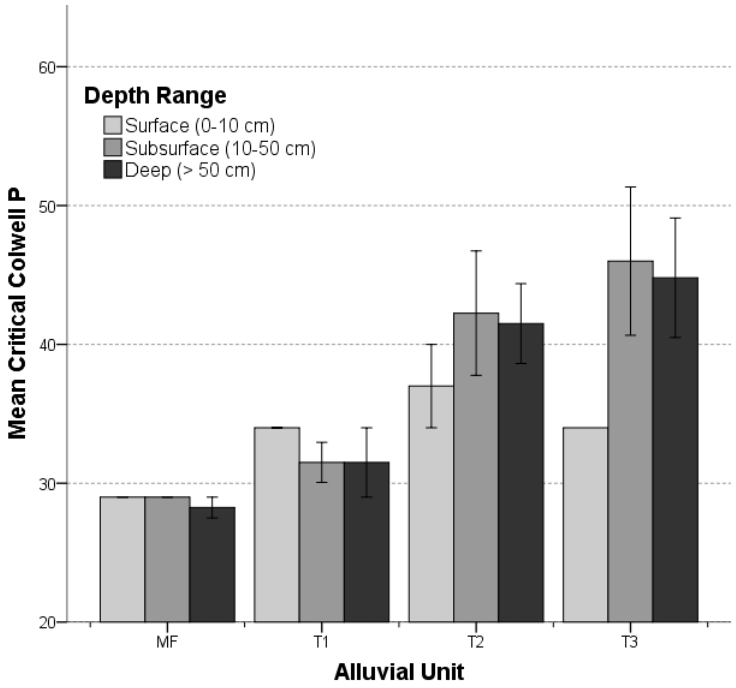

Figure 4.11. Histogram illustrating mean Critical Colwell P values determined for each of the four major alluvial units (Figure 4.1) based on PBI values, determined in accordance with the criteria outlined in Gourley *et al.* (2007) (Chapter 2; Table 2.1). Error bars indicate ±1 standard error value from the mean (refer to appendix XIII for additional information on input data).

4.3.1. Summary

On account of the variations found for PBI between the four major alluvial units, fertiliser requirements are expected to differentiate accordingly, based on the predicted (mean) critical Colwell P values determined for each alluvial unit.

4.4. Dairy A: assessment of overall farm sustainability and risk of phosphorus losses

4.4.1. Effluent treatment practices

Dairy A produces approximately 20, 000 L of dairy effluent per day. The effluent largely consists of residual liquid derived from cleaning processes, i.e., the wash-down of waste accumulated in the milking sheds and holding yards after each milking session. The effluent primarily consists of diluted cattle excreta, but also contains trace amounts of cleaning agents (e.g., acids and alkalies) and milk. Prior to being applied to paddocks, effluent which is produced undergoes a two-stage pond primary treatment process. Effluent applied via irrigation (i.e., from pond two) was tested for key chemical constituents and the results are shown in Table 4.4.

Table 4.4. Effluent chemical analysis results

Parameter	Sample 1	Sample 2	Average
pH	7.45	7.37	7.41
Total phosphorus (mg/L P)	282.3	54.4	168.4
Phosphate (mg/L P)	40.2	53.1	46.6
Salinity (ppt)	1.77	2.52	2.15

Based on information provided by (Dairy A) farm management, rough calculations suggest that approximately 92 L/m^2/year is added within an area of 8.5 hectares (ha), with applications commencing in 1998. According to the results provided in Table 4.4, this means that approximately in excess of 20 tonnes of phosphorus has been added to this area since the commencement of applications. The likely fate of phosphorus added as effluent was assessed by analysing two soil profiles within this 8.5 ha zone receiving regular effluent applications.

Most of the phosphorus derived from effluent treatment was found to be accumulating primarily in the surface layer of the soil profile (Figure 4.12). Contrary to this, however, water extractable phosphorus (WEP) and electrical conductivity (EC) results suggested that phosphorus may be leaching down as far as the deep fraction of the soil profile (Figures 4.13 and 4.14).

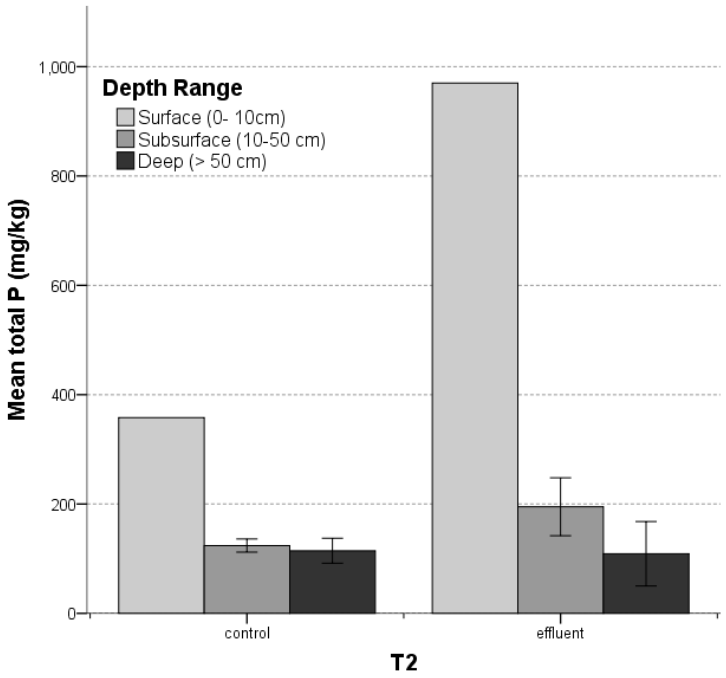

Figure 4.12. Histogram illustrating the mean total phosphorus results determined for T_2 control (AJ-2) soils (left) and effluent treated (EC-1) soils (right) in surface, subsurface and deep depth ranges of the respective soil profiles. Error bars indicate ±1 standard error value from the mean (refer to Appendix IX for additional information regarding the input data).

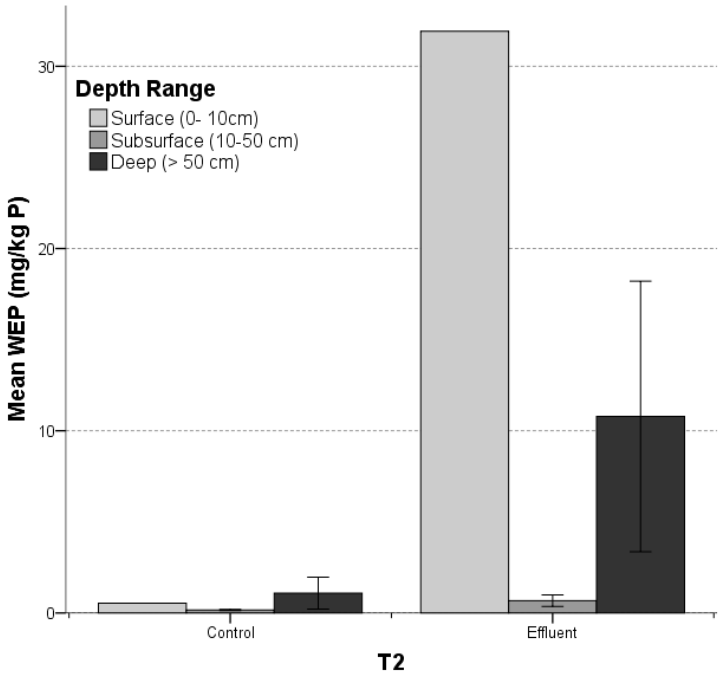

Figure 4.13. Histogram illustrating the mean water extractable phosphorus (WEP) concentrations (mg/kg P) determined for T_2 control (AJ-2) soils (left) and effluent treated (EC-1) soils (right) in surface, subsurface and deep depth ranges of the respective soil profiles. Error bars indicate ±1 standard error value from the mean (refer to Appendix X for further information regarding the input data).

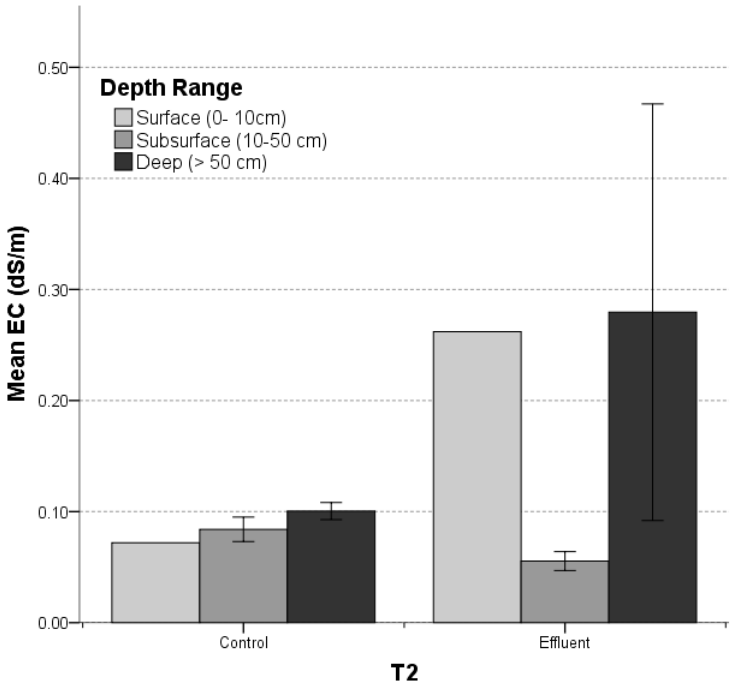

Figure 4.14. Histogram illustrating the mean electrical conductivity (EC) concentrations (dS/m) determined for T_2 control (AJ-2) soils (left) and effluent treated (EC-1) soils (right) in surface, subsurface and deep depth ranges of the respective soil profiles. Error bars indicate ±1 standard error value from the mean (refer to Appendix VIII for additional information regarding the input data).

Statistical methods suggested that increases in WEP with depth were attributable to P derived from effluent leaching down the profile, as EC was found to have a strong positive correlation with WEP in the T_2 effluent soils (r_s = 0.900, p = 0.034), but not for the control soils (r_s = -0.144, p = 0.758) (Appendix XII). In further support of this, both WEP and EC values for surface, subsurface and deep depth ranges displayed a similar pattern when comparing Figures 4.13 and 4.14. Collectively, these results suggest that in effluent treated soils, P is leaching down the profile and accumulating in a deep section (i.e., > 50 cm) of the T_2 effluent treated soil profile.

4.4.2. Current P status of the study site (Dairy A)

Signs of P accumulation varied across the four major alluvial units (Figure 4.15). Cumulative mean total P (including surface, subsoil and deep depth ranges) measurements shown in Figure 4.15 indicate that the majority of P is accumulating in the surface (0-10 cm) of the soil profile, followed but the subsurface (10-50 cm) and then deep (> 50 cm), resulting in total P across all major alluvial units showing a moderately negative

correlation with depth (r_s = -0.624, p = < 0.001; Appendix XIV). Paddocks within which effluent is applied appear to be the least efficient in terms of P fertiliser use, as the values of Cowell P measured are far in excess of the critical Colwell P, respective of each soil type (Figure 4.16; Appendix XIII).

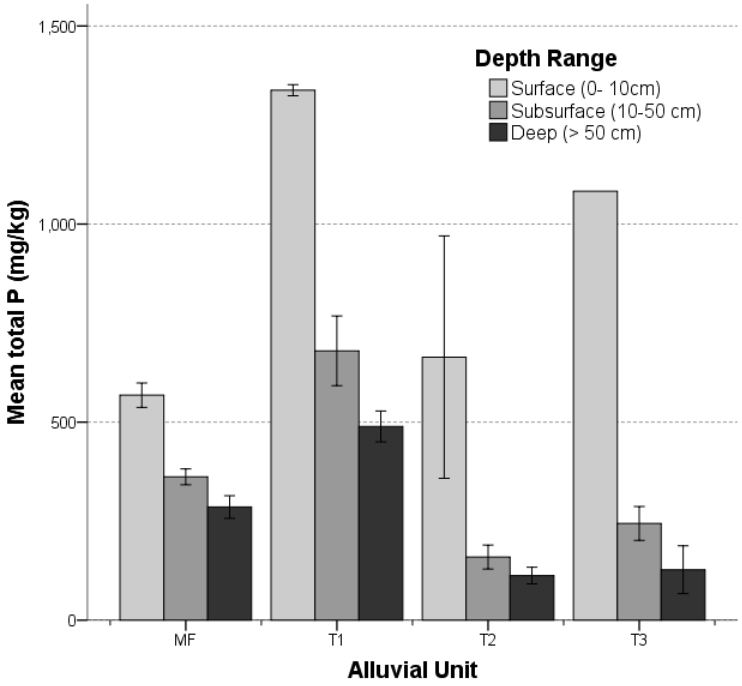

Figure 4.15. Histogram illustrating the mean total phosphorus (P) concentrations determined for surface, subsurface and deep depth ranges of soil profiles sampled from the four major alluvial units (Figure 4.1). Error bars represent ±1 standard error value from the mean (refer to Appendix I for additional information regarding the input data).

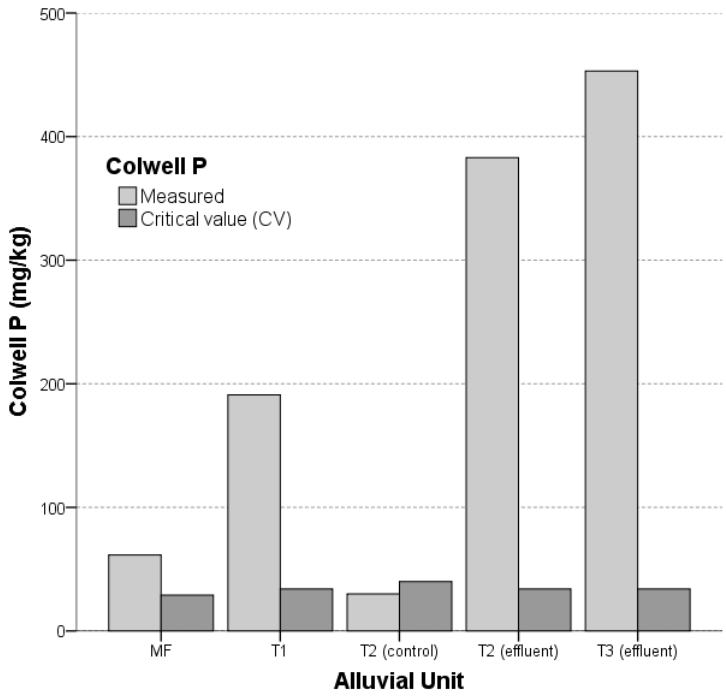

Figure 4.16. Histogram comparing the mean measured Colwell P concentrations found in surface soils (0-10 cm) versus the mean critical value (CV) Colwell P determined for soils from each of the four major alluvial units. For additional information regarding input data refer to Appendix XIII.

Differences in total P concentrations found to exist between major alluvial units occurring within the same paddock may suggest that considerable P losses to the environment (i.e., no longer part of the farm P cycle) are occurring if it is assumed that phosphorus inputs (i.e., via fertiliser) are relatively equal, as the total P is much lower in MF soil profiles (W1 and W2) as compared to the T_1 soil profile (W3) (Figures 4.17 and 4.18). This is particularly true for the surface depth range of the soil profile (Figure 4.18).

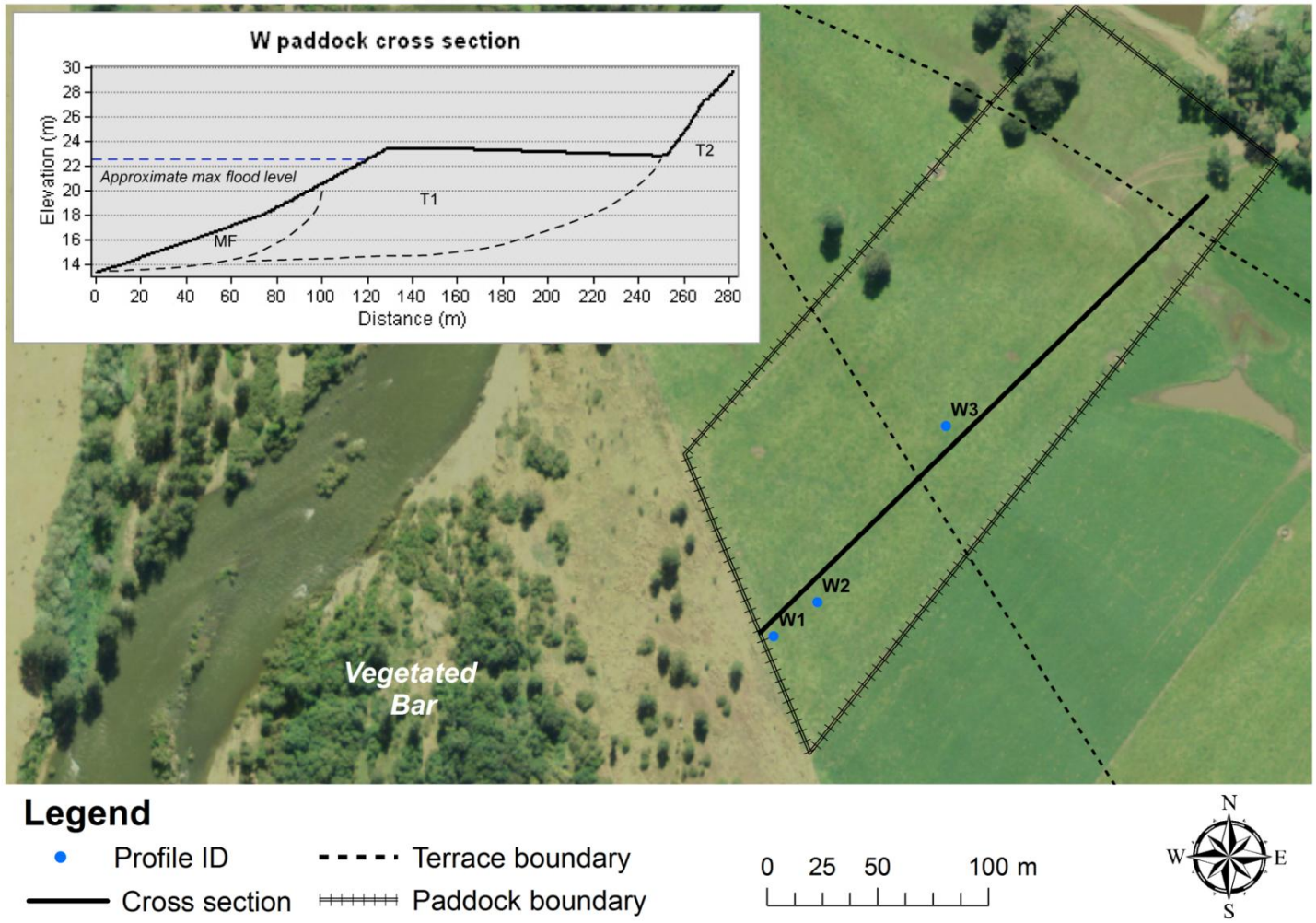

Figure 4.17. Map showing an example where several terraces occur within the same paddock, where: dashed lines represent inferred stratigraphy boundaries. Max flood level approximated from value derived from nearby gauge at Killawarra bridge (site no. 208004) (NSW DPI 2015).

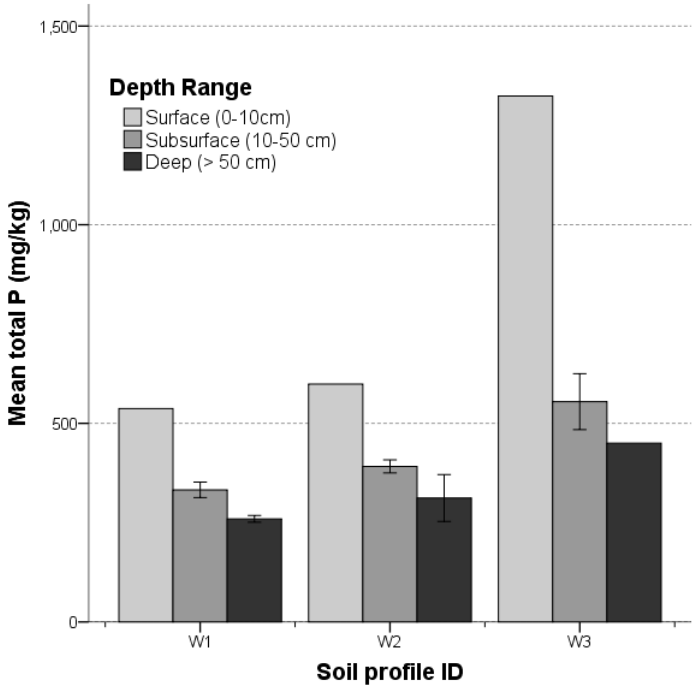

Figure 4.18. Histogram comparing the mean total P concentrations in soil profiles derived from adjacent terraces of variable height within the same paddock (Figure 4.17). Error bars represent ±1 standard error value from the mean (refer to Appendix I for further information regarding the input data).

4.4.3. Summary

P was found to be accumulating predominantly in surface soils for all of the four major alluvial units, with the most severe cases occurring in the effluent treated paddocks. In addition, however, effluent treated paddocks demonstrated signs of WEP accumulation in the deep depth range of the soil profile, suggesting there might be an impermeable layer beneath the solum where AJ-1 and EC-1 samples were taken (see Appendix XVI).

Effluent treated paddocks were found to have the lowest fertiliser use efficiency based on the measured Colwell P values compared with critical Colwell P values. However, the most likely losses of P from the farm system (or cycle) to the environment are expected to be occurring within paddocks along the western side of the study area, where MF, T_1 and perhaps T_3 exist within the same paddock. As mean total P concentrations were lower for two selected MF soil profiles compared to a T_1 soil profile, despite them probably receiving the same inputs of P fertiliser due to being located within the same paddock.

CHAPTER 5

Discussion

Chapter 5 discusses the overall findings of the study. Section 5.1 explains the significance of variations found to exist been between major alluvial units of varying height and how they compare with past research outputs. Section 5.2 highlights and discusses how the identified variations in soil profiles between the modern floodplain alluvial unit(s) and terraces may influence the overall movement and fate of phosphorus (i.e., P dynamics). Section 5.3 explains how knowledge gained from these findings can be put into practice and used to improve P management within this farm system and perhaps others set in similar landscapes. Section 5.4 provides an overview of the overall sustainability of Dairy A based on the overall findings of the study, in light of the current landuse and nutrient (P) management practices.

5.1. Variations in soil properties between major alluvial units

The present study was designed to determine the effect of varying alluvial soil profiles on phosphorus (P) dynamics. An initial objective of the project was to identify whether significant variations between soil profiles were occurring at different locations within the study area - Dairy A. Previous research has shown that variations in soil profiles tend to exist within sequences of stepped alluvial units (Walker 1963, Warner 1972, Walker & Coventry 1976, Walker 1989), as was outlined in Chapter 2, section two. The results revealed that a number of soil profile variations existed between the major alluvial units present within the study area (Chapter 4, section 4.1). In addition, these variations were found to occur in a pattern similar to those which have been previously described for other alluvial landscapes within eastern NSW (Walker & Coventry 1976, Walker 1989). Thus, the first objective of the study was successfully achieved. The variations in soil properties and patterns in which they occurred are outlined and discussed below in the order they were presented in Chapter 4.

5.1.1. Grain size

Collectively, variations in grain size distributions can be recognised to have resulted primarily as a function of two types of mechanisms: (1) Allogenic and (2) Authogenic. Allogenic variations refer to those which have been inherited from fluvial or aeolian depositional processes, whereas authogenic variations include those attributable to *in situ* transformations, resulting in 'profile development' (Leopold *et al.* 1964, Walker 1989). Variations found to exist between the MF and T_1 alluvial units are more likely to have been dominated by allogenic mechanisms, as both alluvial units expressed negligible signs of profile development. Conversely, variations observed between the modern floodplain and T_1 profiles through to the T_3 profile were more indicative of soil profile development being a major differentiating factor. The transition between dominating allogenic to authogenic related variations can be explained by the notion that 'true' profile development does not commence until the alluvial unit has been effectively 'abandoned' by the associated hydrological regime (Chapter 2, section 2.10; Leopold *et al*. 1964, Warner 1972).

Previous studies have shown that rivers may construct multiple modern floodplain alluvial surfaces; these are sometimes confused to have different basal ages as a result of age-height correlation based inferences (Walker & Coventry 1976, Cohen & Nanson 2008). Both the MF and T_1 soil profiles showed minimal evidence of pedogenic development (as will be explained further below). Before profile development can commence, it is required that the floodplain is left undisturbed by the associated (active) hydrological regime for a significant amount of time (Walker & Coventry 1976, Cohen & Nanson 2008). Consequently, even though the surfaces of the MF and T_1 were located at different heights above the existing channel, it is possible that these units may both be of the same contemporary alluvial unit, reflecting two modern floodplain surfaces; i.e., the low and high floodplain (Warner 1992).

A number of past studies have revealed that soil profiles from adjacent low and high floodplain surfaces are often characteristically dissimilar, despite being considered as one characteristically defined alluvial unit (i.e., the modern floodplain) based on active hydrological associations (Warner 1972, Cohen & Nanson 2008). This explanation was found to be applicable for the current study area, as soil profile differences found to occur between MF and T_1 soils were typical of those described in the literature (Cohen &

Nanson 2008). For example, the T_1 soil profiles consisted predominantly of finer-grained, horizontally laminated alluvium reflecting an upward fining sequence built by vertical accretion; whereas, the MF soil profiles were more heavily dominated by courser-grained alluvium, with sporadic seams of sand and gravel throughout the profile, indicating a cut-and-fill type of construction (Appendix I). Hence, both sand and mean grain size results were found to be higher within MF soils as compared soils from the T_1 unit (Chapter 4, Table 4.1, Figures 4.3 and 4.4). Collectively, these findings further demonstrate that variations in soil profiles between the MF and T_1 units may be recognised as being dominated by allogenic mechanisms. In addition, these results also suggest that soil profiles from the two lowest alluvial units investigated (MF and T_1) may represent the first two common soil profile stages seen to occur in alluvial landscapes, as described by Walker & Coventry (1976) and later illustrated by Walker (1989). Respectively, these are, *stratic* (MF) and *cumulic* (T_1) (Chapter 2, section 2.10; Figure 2.14).

The MF unit may be recognised to encompass characteristics from both *stratic* and *cumulic* stages; thus, it should not be ruled out that MF and T_1 units may both represent the *cumulic* stage (Walker & Coventry 1976). Similarly, according to Warner (1972), the MF and T_1 units may also be referred to as the (contemporary) *floodplain* and *unaltered terrace* (Chapter 2, section 2.11). According to these interpretations, it is possible that the T_1 unit represents a 'recently abandoned' terrace; indicating that limited profile development is attributable to recent (i.e., during the Holocene), or at most, seldom inundation by the currently active hydrological regime (Warner 1972). Nonetheless, irrespective of which explanation is used, these findings suggest that soil profiles found within the MF unit are much more likely to display higher expressions of local variations within soil profiles due to inconsistencies in the 'unmodified' stratification of alluvium (Iqbal *et al.* 2005), as compared to the other terraces (T_1, T_2 and T_3) which are no longer actively associated with the current hydrological regime – i.e., abandoned (Warner 1972, Walker & Coventry 1976).

Variations in grain size found to occur between terraces (T_1, T_2 and T_3) are likely to be primarily attributable to profile development. For example, illuviation of *in situ* clay may be used to explain why higher (older) terrace (T_2 and T_3) soils were found to have greater cumulative clay contents below the surface depth range, as compared to the MF and T_1 soil profiles (Chapter 4, Table 4.1 and Figure 4.2) (Brewer & Walker 1969). Illuviation

essentially refers to the movement and accumulation of clay as a function of weathering and pedological processes coupled with time (Leopold *et al.* 1964). Hence, illuviation of clay is considered to be one of the most fundamental outcomes of soil profile development (Brewer & Walker 1969, Chittleborough & Walker 1988).

It is possible to approximate the extent to which a soil profile has undergone profile development through determining the amount of clay formed *in situ* relative to the 'total' clay, which includes additional clay particles formed elsewhere that have been subsequently transported and deposited via fluvial or aeolian processes (Brewer & Walker 1969). Additionally, comparing the various size fractions of clay (e.g., < 0.03 µm, 0.2-0.06 µm and 2-0.2 µm) can also provide an indication of the extent of profile development, as clay size distributions tend to become finer with time (Chittleborough & Walker 1988, Walker 1989). Determining these parameters would, however, require additional methods which were considered to be beyond the scope of the study, given the time and resources available. Alternatively, an estimation of clay illuviation was made by comparing the calculated ratio of the maximum clay content in the subsoil (> 10 cm deep) to the minimum clay content measured in the surface soil (0-10 cm deep), similar to the method outlined in Walker (1970), denoted as the "degree of particle size differentiation". For this particular method, higher ratio values indicate that soils have undergone a greater extent of soil development. The degree of particle size differentiation calculated for each major alluvial unit revealed a pattern which was in accordance with notions regarding age-height correlations; hence, indicating increasing soil development from the modern floodplain through to T_3 (Walker 1970).

The boundary between surface clay and subsurface clay was much more prominent for the two highest alluvial units (T_2 and T_3), although, perhaps not sharp enough to be termed a duplex soil (Chittleborough & Cox 1999). Mean grain size and sand content tended to decrease in soil profiles (all depths) from the MF through to T_2, which was similar to T_3. Hence, sand and mean grain size expressed a similar pattern of variation with location.

Weathering processes generally result in coarser material such as sand and gravels to break down into finer particles, which are ultimately converted into silt and clay (Summerfield 1991, vanLoon & Duffy 2010). In addition, deposition of fine-grained aeolian material (e.g., dust) may also contribute to the silt and clay fraction of soils within alluvial units (i.e., loess), with accumulation becoming greater following abandonment

(Eger *et al.* 2012). As a result, the youngest, most recently deposited alluvium generally has higher sand and mean grain size contents, as these soils have had relatively little time (since deposition) to be altered via subsequent weathering processes and concomitant aeolian accession (Warner 1972, Chartres *et al.* 1988, Eger *et al.* 2012). Mean grain size was found to increase with depth for the MF and T_1 profiles, this was largely expected due to them both being more recent alluvial deposits, which often demonstrate an upward fining sequence (Cohen & Nanson 2008). Interestingly, mean grain size results indicated a similar trend with depth for T_3 soils, but, in this case more likely reflecting an abrupt transition from thick illuvial clay (small mean grain size) to the weathered deposit (courser material) (Walker & Coventry 1976). Unlike the other locations (MF, T_1 and T_3), sand and mean grain size decreased with depth in T_2 soil profiles, although this trend may have potentially been reversed if sampled deeper.

Variations pertaining to grain size occurred predominantly in the subsoil (i.e., subsurface and deep) depth ranges of soil profiles, with surface soil grain sizes showing little differentiation with location (Appendix III). The relatively small degree of variation in grain size results determined for surface soils may be due to a number of factors. Firstly, recent contributions of aeolian material (e.g., silt and clay) to surface soils may be relatively equal, resulting in similar grain size characteristics. Secondly, surface soils are all exposed to relatively similar weathering environments and landuse practices. Finally, anthropogenic disturbances (e.g., tillage and irrigation) may have somewhat altered the occurrence of natural variations by inducing physical and chemical weathering (Iqbal *et al.* 2005, Vaananen *et al.* 2008, Simmonds *et al.* 2015).

5.1.2. Mineralogy

Clay mineralogy varied considerably between terraces in a succession which was indicative of profile development occurring in proportion to age-height based correlations. Firstly, the cumulative abundance of minerals associated with P sorption (i.e., illite and kaolin) in soil profiles increased progressively from the lowest (MF) through to the highest (T_3) alluvial unit (Chapter 4, Figure 4.6). This is likely to have resulted from differing extents of weathering processes exerted between younger versus older soil profiles; hence, profile development is likely to be the main contributing factor to the variations in mineralogy found to exist in the sequence of major alluvial units investigated

(Brewer & Walker 1969). This notion, however, is based on the assumption that the nature of deposited alluvium was relatively similar across all terraces at the time of deposition. This may be considered a reasonable assumption to make, as the types of primary weathering minerals (i.e., albite, orthoclase and labradorite) and more notably, quartz content, were found to be relatively constant among all terraces (Appendix II). In most environments, quartz is much less affected by prolonged weathering, thus, might be used as a crude indicator that the pattern of clay mineralogy is most likely to have resulted from profile development (Summerfield 1991).

Weathering of parent material to form secondary minerals often occurs as a progression (Figure 5.1); thus, in certain cases, the sequences and relative abundances of clay minerals may reflect a temporal succession (Summerfield 1991). This explains why progressive increases in the amounts of illite, kaolin, goethite and gibbsite were found between the MF through to T_3 terrace soils, as this sequence (tentatively) represents the youngest to oldest alluvial units (Chapter 4, Figure 4.6).

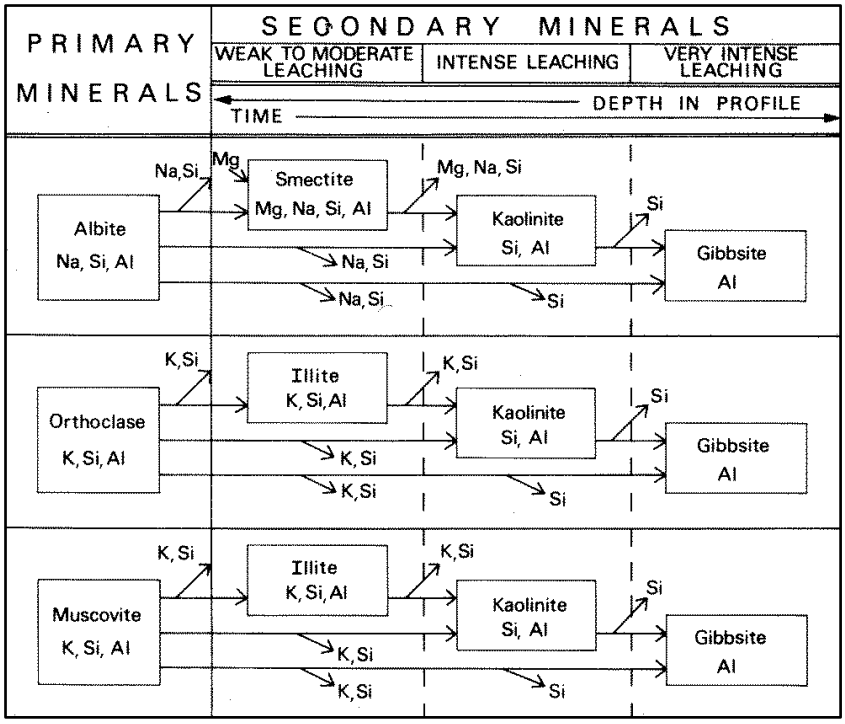

Figure 5.1. Examples of progressive transformations in mineralogy from parent material through to secondary products (adapted from Summerfield 1991, pg. 140).

The transition from illite to kaolin dominated soils was demonstrated from the T_1 through to the T_3 terrace (Appendix II). Kaolinite is generally formed by a much slower rate of weathering compared to illite (Leopold *et al.* 1964). Hence, soils dominated by kaolin are often expected to be older than those dominated by illite (Brewer & Walker 1969, Chittleborough & Walker 1988, Walker 1989). This can be can be illustrated by the following sequence (also shown in Figure 5.1), which typically occurs in most weathering systems (Leopold *et al.* 1964):

Orthoclase -> Illite -> Kaolinte -> Gibbsite

Hence, the ratio of illite to kaolin in a soil can provide an indication of the extent of profile development. Similarly, goethite and gibbsite may also occur as secondary minerals occurring from weathering processes (Figure 5.1); hence, their abundance may also be used to indicate variations in profile development between different locations. It is worth noting, however, that time is not the only factor which determines the outcome of weathering, as varying rates of (a) removal by erosion, (b) influences of fluctuating environmental conditions, and, to a lesser extent (c) anthropogenic influences may also play a role (Leopold *et al.* 1964, Summerfield 1991, Huggett 1998). Thus, although parent material was assumed to be common among the four major alluvial units (MF, T_1, T_2 and T_3), other factors (e.g., climate, vegetation and topography) may have also been altered throughout time (Huggett 1998). Consequently, time alone cannot be used to explain the present outcome of weathering. Nonetheless, the general patterns of profile development indicated by mineralogy results which were found to occur in the assemblage of stepped alluvial units in this study were similar to those revealed in previous studies, all of which suggested time as being the dominant differentiating factor (Brewer & Walker 1969, Warner 1972, Walker & Coventry 1976, Walker & Hutka 1979).

5.1.3. Aluminium and Iron Oxyhydroxides

Iron and aluminium oxyhydroxides may also occur as secondary products of weathering (Summerfield 1991). Hence, the abundance of Al_{ox} and Fe_{ox} structures may be influenced by weathering processes and profile development. The cumulative abundance of Al_{ox} in soil profiles (surface, subsurface and deep) were found to increase progressively from the MF through to T_3, however, this pattern was not apparent for Fe_{ox} (results section 4.1.3; Figures 4.7 and 4.8, respectively). Several explanations are possible as to why this was the

case. Firstly, the crystallinity of Fe mineral structures might be being altered in some locations, which is causing the Fe_{ox} results to display an unexpected pattern. Crystallinity of Al and Fe oxyhydroxide structures may be influenced by factors such as OM, pH and redox conditions, which may in some cases, render them unquantifiable by the (acid ammonium oxalate extractable Al and Fe) method used in his study (Rayment & Lyons 2011, Coad *et al.* 2014). This is because this method is designed to quantify the 'active' proportion of Al and Fe oxyhydroxides only, meaning that Al and Fe recovered from the extraction procedure represent those derived from predominantly short-range-order (poorly) crystalline structures (Rayment & Lyons 2011). This is not to say that Al is not also being effected, but, it has been previously shown that the extraction method may sometimes be more specific in extracting poorly crystalline forms of Fe than for Al; thus, perhaps resulting in the determined Al_{ox} concentrations being disproportionately influenced by more crystalline structures (Singh & Gilkes 1991). Secondly, organic matter may be influencing the abundance of Fe_{ox}, as Fe oxyhydroxide coatings are generally associated with the surface of organic material (Coad *et al.* 2014). This might also explain why Fe_{ox} concentrations tended to be higher in surface soils, as surface soils contained more organic matter than the two deeper depth ranges (i.e., subsurface and deep) in all soil profiles analysed (Appendix I). Another possible reasoning for the unexpected pattern of Fe_{ox} concentrations may be explained by natural variations in the soil mineralogy (Yaghi & Hartikainen 2013, Abdala *et al.* 2015). Nevertheless, the point to highlight here is that natural presences of Al_{ox} and Fe_{ox} are likely to increase in conjunction with age-height correlations found in stepped alluvial assemblages; even though the results from this study could only confirm that this was the case for Al_{ox}, but not for Fe_{ox}.

Despite the fact that no statistically significant correlation could be revealed between Al_{ox} and depth, it was apparent that Al_{ox} did tend to be higher in the subsurface and deep depth ranges of soil profiles (Chapter 4, Figure 4.7). This pattern may be partly explained by the fact that Al is the main octahedral cation in clays (Yaghi & Hartikainen 2013), and as a result, total Al is likely to express a similar pattern with depth as clay, which was found to increase with depth in most soil profiles. In contrast, however, it could be argued that this explanation is potentially invalid, as the reagent used in the extraction method is supposed to have a minimal effect on clay; hence rendering clay associated Al ions largely insoluble (Rayment & Lyons 2011). Moreover, this pattern of Al_{ox} distribution with depth may result from Al_{ox} derived from overlying elluvial horizons leaching down the profile and

accumulating in the illuvial soil horizons making up the bulk of the subsoil profile (Almond & Tonkin 1999, Eger *et al.* 2012).

Fe_{ox} concentrations tended to display an inverse pattern to that described for Al_{ox}; where, Fe_{ox} concentrations were mostly higher in surface soils and generally decreased with depth (Chapter 4, Figure 4.8). Once again, this may be occurring as Fe and Al concentrations determined by the ammonium oxalate extractable method may not necessarily reflect equal proportions of 'active' Al and Fe (non)crystalline structures to the total amount of Al and Fe oxyhydroxides present (Singh & Gilkes 1991). In addition, it has previously been shown that Al tends to be more mobile in soils compared to Fe, hence resulting in patterns of Fe_{ox} concentrations demonstrating minimal signs of Fe illuviation (Almond & Tonkin 1999).

5.1.4. Summary

The parameters discussed above were chosen for two important reasons: (1) due to their potential to influence P dynamics and (2) because of all parameters known to influence P dynamics, these are the least likely to be significantly affected by anthropogenic disturbance as a result of farm management practices (particularly in the subsoil). This allows for inferences derived from the study to be applicable in other similar landscapes, irrespective of variations in land management practices. It was revealed that the concept of increasing soil development with increasing landscape age (inferred from height above the existing channel) was demonstrated by the sequence of major alluvial units examined in this study. This pattern has been revealed in numerous previous studies for similar landscapes throughout eastern NSW. Thus, the following (section 5.2) findings from the study are expected to be applicable not only on a local, but potentially at a regional scale.

5.2. Influence of soil profile variations on phosphorus dynamics

Alluvial landscapes characterised by a stepped sequence of terrace and modern floodplain units abutting the bedrock valley margin represent a common landform assemblage found within many parts of eastern NSW (Warner 1972, Walker & Coventry 1976). Alluvial landscapes are often associated with agronomic industry due to their inherently nutrient rich soils in conjunction with water accessibility (Rinklebe *et al.* 2007). Prior studies have highlighted the influence that certain soil properties can have on P dynamics, including those discussed in the previous section; i.e., grain size, mineralogy and Al_{ox} and Fe_{ox} (Chapter 2, section 2.5). Therefore, one of the major aims of this project was to make an attempt to build upon previous research carried out on stepped alluvial landscapes in eastern NSW by linking geomorphic features with nutrient (i.e., P) dynamics. Thus, the second major objective of the study was to assess the influence of variations in grain size, mineralogy and, Al_{ox} and Fe_{ox} on the movement and fate of soil P following fertiliser application between alluvial units of alternating height above the existing channel.

Soil P dynamics are primarily controlled by two key factors: (1) movement pathways (Maguire & Sims 2002) and (2) sorption capacity of the soil (Holford *et al.* 1997, Chittleborough & Cox 1999). Both of these factors have been shown to be largely influenced by the key parameters outlined in section 5.1 (Chapter 2, sections 2.5 & 2.8).

Phosphorus generally has a high affinity for soil, which often results in the limited mobility of P in the solum (Chittleborough & Cox 1999, Abdala *et al.* 2015). This is largely attributable to P sorption reactions which occur between labile P and soil particles (Chapter 2, section 2.4.3). Traditionally, the sorption capacity of a soil has been determined by generating P sorption isotherms; although, this is often regarded as an expensive and time consuming task (Pöthig *et al.* 2010). Thus, in more recent years, researchers have worked on developing time-efficient, inexpensive methods for determining a soils sorption capacity more suited to commercial laboratory applications (Burkitt *et al.* 2002). An example of one of these types of methods is the phosphorus buffering index (PBI): a single point P sorption method which is now considered the national standard test for estimating soil sorption capacities (Gourley *et al.* 2007a, Weaver & Wong 2011). Grain size distribution, mineralogy and Al_{ox} and Fe_{ox} have all been shown

to influence P sorption, and hence, the PBI in soils (Chapter 2, section 2.5; Coad *et al*. 2014).

The influence of P sorption on the movement of P is most dominant in soils with low P concentrations and high PBI (Weaver & Wong 2011). When PBI is high, P mobility is limited unless there are available preferential flow paths with minimal sorption potential, providing avenues through which P can move freely (Hesketh & Brookes 2000, Ahmed *et al.* 2008, Weaver & Wong 2011). Conversely, once the phosphorus buffering capacity nears exhaustion, P movement becomes less restricted by sorption processes; thus, when PBI is low and phosphorus is high, P mobility is enhanced (Ahmed *et al.* 2008, Pöthig *et al.* 2010).

Collectively, the nature and sequence of variations in soil profiles between terraces resulted in a trend which reflected a pattern whereby the cumulative (i.e., surface, subsurface and deep) PBI in soil profiles progressively increased from the lowest (MF) to highest (T_3) terrace (Chapter 4, Figure 4.9). When comparing individual depth ranges, however, this trend was only statistically significant for the subsurface and deep depth ranges of the soil profile, whereas surface soils showed only minimal variation (Chapter 4, section 4.2; Appendix III). This pattern is likely to be primarily attributable to variations in clay content and Al_{ox}, which were found to be positively correlated with PBI, as well as decreasing sand content, found to be negatively correlated with PBI (Chapter 4, section 4.2). This inference is likely to be valid according to the abundance of supporting literature which has revealed that Al_{ox} and clay content play a major role in enhancing a soils sorption capacity (Chapter 2, sections 2.5.1 & 2.5.4), and that higher sand content generally reduces the ability of a soil to sorb P (Lewis *et al.* 1981, Whalen & Chang 2001, Ahmed *et al.* 2008). In addition, sand has also been linked with enhancing desorption of soil-P (Singh & Gilkes 1991). Nonetheless, the important point to highlight here is that with decreasing PBI, additions of P are able to move more freely within the soil matrix, which may ultimately result in P being lost to receiving water bodies (Simpson *et al.* 2014). Figure 5.2 provides an illustration of these findings in light of the existing literature pertaining to variations in the stages of profile development common in stepped alluvial landscapes through eastern NSW.

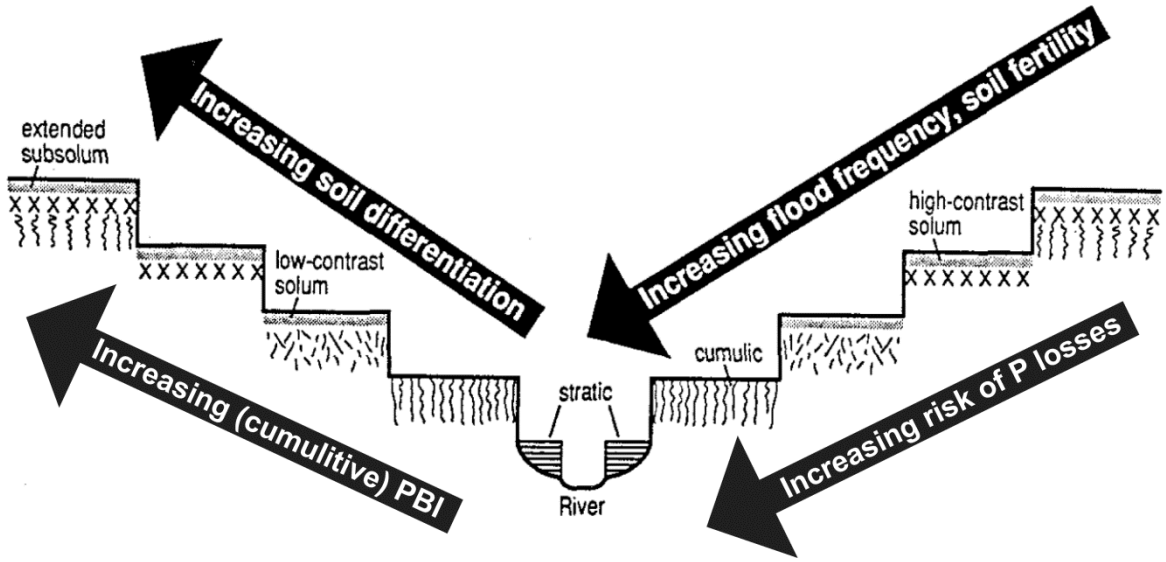

Figure 5.2. Conceptual illustration of the general sequence of soil types present within assemblages of stepped terrace and modern floodplain alluvial units common throughout eastern NSW, including associated progressive changes in (cumulative surface, subsurface and deep) soil PBI and estimated risk of P losses (modified from Walker 1989).

Clay content also plays a major role in influencing phosphorus movement pathways through restricting, and/or, redirecting the movement of the soil solution (Chittleborough & Cox 1999). Soils rich in clay are highly porous (but with small pore diameters) due to the very small particle size of clay constituents and as a result, the soil solution is often more likely to be retained rather than pass through (vanLoon & Duffy 2010). Thus, clay rich soils may be recognised to effectively inhibit the movement of the soil solution, in which large proportions of P may be contained. Soils within the higher terrace units (T_2 and T_3) had notably higher clay contents than the lower two alluvial units (MF and T_1), particularly within the subsurface and deep depth ranges of soil profiles (Chapter 4, section 4.1.1; Table 4.1 and Figure 4.2). This suggests that P added to T_2 and T_3 soils is more likely to be translocated laterally rather than leaching down the profile, as clay would largely inhibit the latter to occur, as a result of enhancing soil sorption potential in conjunction within limiting the mobility of the soil solution (Chittleborough & Cox 1999).

Contrastingly, P added to the MF and T_1 soils is much more susceptible to being leached compared to T_2 and T_3 soils, due to the lower content of clay in the subsoil (i.e., subsurface and deep depth ranges). In addition, increasing sand and mean grain size would also promote downward movement of the soil solution, through providing greater porespace within soil profiles and minimising sites for sorption (Miller *et al.* 2002,

Agudelo *et al.* 2011). Texture contrast between soil horizons may also play a major role in P movement dynamics (Chittleborough & Cox 1999). The boundary between surface and subsurface clay content was considerably higher in T_2 and T_3 soil profiles compared to soil profiles from the MF and T_1 units (Chapter 4, Figure 4.2). This indicates that the losses of P to receiving water bodies from T_2 and T_3 profiles are likely to result from surface runoff rather than leaching, with the opposite being true for T_1 and MF soils. Despite this, it should still be recognised that MF and T_1 soils are also likely to be susceptible to surface runoff (as most soils are), but, perhaps not as much as the two highest terrace soils (T_2 and T_3). For example, when intense downpours of rain occur within a short duration, P may not have a chance to leach into the soil profile, and may consequently be translocated via runoff in dissolved or particulate form (Sharpley *et al.* 2001, McDowell 2012, Adams *et al.* 2014).

Furthermore, typical modern floodplain soils are inherently more variable than terrace units, simply because they are composed of more recently deposited alluvia and have incurred minimal profile development; for this reason, the point of change between soil characteristics with depth can be somewhat arbitrary in modern floodplain alluvial units (Walker 1963, Iqbal *et al.* 2005). For example, seams of sand rich alluvium found between finer grained deposits derived from overbank deposits, deposited during periods of intense flooding. Alluvial stratification features such as this may facilitate the translocation of P by providing preferential flow pathways through highly permeable, low sorbing material, i.e., sand (Sharpley *et al.* 2001, Ahmed *et al.* 2008). Hence, P losses in typical modern floodplain soils may not only be enhanced but also much harder to predict due to more frequent localised variations.

5.2.1. Summary

Phosphorus dynamics varied between terraces as a function of differentiating soil profile characteristics; in particular, clay content, Al_{ox} and sand content. These findings suggest that the varying susceptibility of P losses (to a receiving water bodies) to occur following additions of fertiliser between the four major alluvial units examined can be expected to occur in the following sequence:

$$MF > T_1 > T_2 \geq T_3$$

Figure 5.3 provides a conceptual illustration of findings discussed in section 5.2.

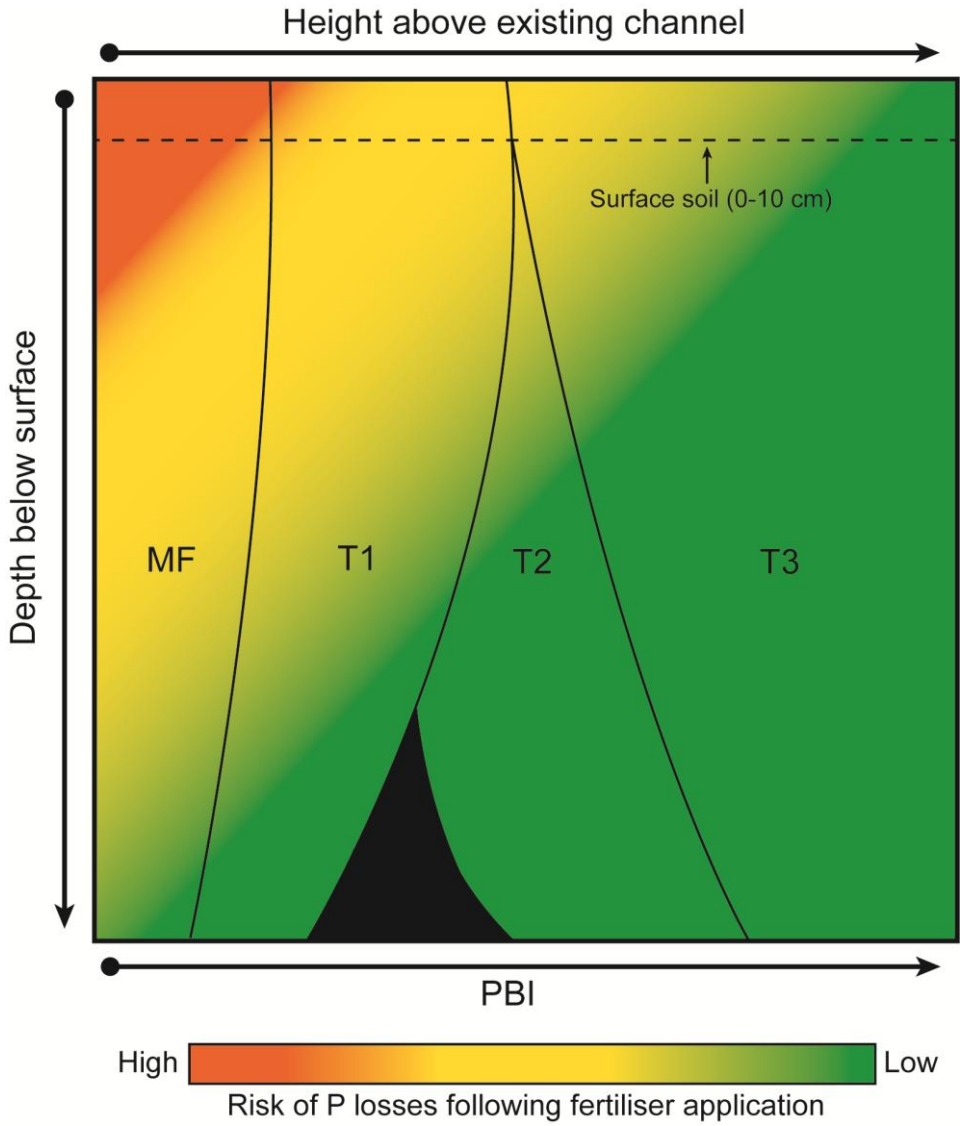

Figure 5.3. Conceptual soil P-loss risk matrix illustrating the varying susceptibility of farm P losses to the environment between the four major alluvial units examined following additions of fertiliser. Note, lines dividing the four major alluvial units represent the changing pattern of PBI with depth shown in Chapter 4, Figure 4.9.

5.3. Implications for improving nutrient management strategies

Findings discussed in sections 5.1 and 5.2 suggest that patterns of varying soil characteristics and hence phosphorus dynamics are likely to be similar for other alluvial landscapes characterised by an assemblage of stepped modern floodplain and terrace units, shown to be common throughout eastern NSW (Walker & Coventry 1976, Walker 1989). A major focus of the present study was to reveal findings that are identifiable on a local scale but potentially applicable on a much wider basis, an important component which other related (river terrace landscape) studies in the past have arguably struggled to achieve (Bowler 1967, Warner 1972, Cohen & Nanson 2008). Therefore, the third major objective of the study was to make an attempt to achieve this by identifying the implications of findings for improving nutrient management strategies not only for the farm under study (i.e., Dairy A), but perhaps others operating in similar landscapes.

The variations in soil profile characteristics identified between the four major alluvial units examined in this study may indicate that considerable fertiliser use inefficiencies are occurring within paddocks where two or more of these units occur. Consequently, major alluvial units of varying heights within stepped alluvial landscapes may represent zones of variable production; where production levels differentiate despite receiving equal inputs of fertiliser. For this reason, it may be appropriate for major alluvial units within these types of landscapes to be considered separate 'management zones' (Hedley 2015). Therefore, paddocks with more than one major alluvial unit may require variable rates of fertiliser, and perhaps other treatments as well (e.g., water via irrigation). This may help to improve overall farm sustainability through optimising fertiliser use efficiency whilst also minimising losses of phosphorus to the environment (Weaver & Wong 2011, Hedley 2015). In addition to environmental benefits, the financial gains from enhancing fertiliser use efficiency provides further impetus for farm managers to improve their current P fertiliser management strategies (Weaver & Wong 2011, Simpson *et al.* 2014, Hedley 2015).

Farm managers now have the ability to apply fertiliser at more optimal rates at the sub-paddock scale through the advent of increasingly available farm machinery equipped with sophisticated GIS technologies (McKinion *et al.* 2001, Simpson *et al.* 2014, Hedley 2015). However, the costs involved with implementing these advancements remains a major

limitation, despite the potential for long term capital gains (Simpson *et al.* 2014, Hedley 2015). The findings from this study suggest that determining variable rates of fertiliser based on major changes of relief may provide a cheaper alternative, as these areas are easy to differentiate, therefore, negating the need for GIS technologies. In addition, this research may make farm managers more aware of the inter-paddock variations likely to exist as a result of having one or more major alluvial units of varying heights, thereby potentially influencing future decisions pertaining to routine agronomic testing methods; for example, determining where (and perhaps at what depths) samples collected for analysis are taken from within individual paddocks.

It is worth stressing again at this point that the pattern of PBI values found within the sequence of stepped alluvial units examined in this study was primarily attributable to the subsurface and deep fractions, whereas mean surface soil PBI values showed no significant difference between the four major alluvial units (Chapter 4, section 4.1.4; Appendix III). This is an important finding of the study, as it may suggest that common methods used for routine agronomic soil testing (primarily conducted in order to predict fertiliser requirements) do not account for subsoil variations, which may result in fertiliser use inefficiencies as well as an increased risk of P losses.

Traditional agronomic testing methods designed for grazed pasture systems generally only require the top 10-15 cm of the soil profile to be sampled for analysis; thus, nutrient management strategies rarely take into account subsoil conditions (Sharpley *et al.* 1996, Rayment & Lyons 2011). Restricting sampling methods to surface soils, however, limits the ability for agronomic tests to be utilised for predicting subsoil P dynamics and associated loss pathways (Torbert *et al.* 2002, Rayment & Lyons 2011). Thus, if current agronomic sampling methods were to be used for environmental related purposes, data obtained may provide results that are unrepresentative of the actual risk of P losses to occur if taking subsoil variations into consideration. For example, surface soil results in this study suggest that PBI values between the four major alluvial units might be assumed to be relatively constant; however, this was shown to be incorrect when taking subsurface and deep soils into account. One might argue that measuring the PBI in subsoils is financially unviable, as plant-nutrient-soil interactions tend to progressively diminish with depth (Rayment & Lyons 2011). However, from an environmental perspective, this

knowledge is critical in being able to assess the potential for subsurface losses to occur; for example, via leaching or subsurface lateral translocation.

Recent research efforts have investigated certain soil tests which may be beneficial for both agronomic and environmental based assessments (Wang *et al.* 2015). These methods, however, don't usually incorporate subsoil testing, and are primarily designed to provide an environmental assessment tool exclusively used to predict potential P losses via runoff (Wang *et al.* 2007, Wang *et al.* 2012). Thus, findings from this study suggest that future research should now aim to develop more tests which can be used to predict losses of P via subsurface routes, whilst also having useful agronomic implications as well. The PBI test may be recognised as an existing method which fits the aforementioned criteria and should, therefore, be encouraged to be more frequently tested in subsoils.

In a number of studies, PBI has been strongly linked with nutrient availability, whereby, the higher the PBI value, the more nutrients are likely to be required for optimal production (Burkitt *et al.* 2002, Gourley *et al.* 2007a, Moody 2007). As a result, PBI test values have recently been used to determine suitable ranges of extractable soil P concentrations (e.g., Colwell P) needed in soils to achieve 90-95% maximum pasture production (Gourley *et al.* 2007a, Weaver & Wong 2011, Simpson *et al.* 2014). The concentrations determined from this method are frequently referred to as critical (Colwell) P values (Simpson *et al.* 2014).

According to the pattern in PBI values determined in this study, the four major alluvial units examined are likely to require different inputs of fertiliser, on account of each major unit having different associated critical Colwell P values (Chapter 4, Figure 4.11). This is likely to occur in other stepped alluvial landscapes as well, as the pattern of variations in soil characteristics between major alluvial units (i.e., modern floodplain and stepped terraces) found to be influencing the PBI have shown to be frequently common in eastern NSW (Walker & Coventry 1976). If two or more terraces occur in one paddock, it is likely that at least one will receive either more or less phosphorus fertiliser than is required for optimal production, as single paddocks are often treated with a relatively even distribution of fertiliser. Hence, this may be recognised as a significant finding of the study.

Surplus P concentrations may be quantified (approximately) by comparing the calculated critical Colwell P and the measured Colwell P, as was shown in the results (Chapter 4,

Figure 4.16; Appendix XIII). Similarly, this method can also be used to determine the required inputs of fertiliser (Weaver & Wong 2011). As a result, comparing the measured Colwell P to the determined critical Colwell P is commonly used for assessing P fertiliser use efficiency throughout Australia (Gourley *et al.* 2007a, Weaver & Wong 2011, Simpson *et al.* 2014). For instance, an equal value for both parameters would reflect an optimal level of soil phosphorus, thereby, signifying efficient fertiliser application rates (Simpson *et al.* 2014). Ideally, plant available P (i.e., Colwell P) should not exceed the requirements of the receiving crop or pasture, but, this balance may be difficult to achieve when determining appropriate fertiliser inputs due the need to account for pre-empted losses of P to products, runoff and leaching, which may be hard to predict (Weaver & Wong 2011, Simpson *et al.* 2014).

5.3.1. Summary

It was revealed that soil profiles of major alluvial units of variable height within stepped alluvial landscapes are likely to require variable inputs of P fertiliser in order to achieve maximum production levels. Accordingly, it may be beneficial to amend methods of soil sample collection for agronomic testing in paddocks where several major alluvial units occur. For example, by collecting samples which are representative of alluvial units differentiated by height above the existing channel, rather than for an entire individual paddock. Furthermore, this research may be used to help improve farm design, so that paddocks can be set out a way as to eliminate the possibility of more than one major alluvial units occurring in the same paddock. Thus, findings from this study may provide a major step towards helping farms achieve more sustainably practices through understanding how soil characteristics are likely to vary within stepped alluvial landscapes, which may help improve (a) fertiliser use efficiency, (b) agronomic soil test sampling techniques and (c) farm design.

5.4. Dairy A: sustainability and risk of phosphorus losses

Dairy A is one of the largest dairies currently operating along the Manning River. As a result, local water catchment authorities have demonstrated an interest in assessing the potential for nutrient contamination of water resources to be contributed from Dairy A. Therefore, the fourth major objective of the study was to make an assessment of the current sustainability of the present operations of Dairy A and the potential risk for P losses (from the farm system) to be occurring, with a special focus on effluent treatment practices.

Phosphorus levels in surface soils were higher than the predicted required levels for maximum growth based on measured Colwell P concentrations compared with calculated critical Colwell P values (Chapter 4, Figure 4.16). This suggests that runoff may be of a concern, particularly for T_1 soils (Chapter 4, Figure 4.16), as WEP was found to be positively correlated with Colwell P (Chapter 4, Table 4.2). In addition, according to the current Colwell P concentrations in surface soils, further additions of P fertiliser (perhaps excluding T_2 control soils) would have minimal benefits on yield (Chapter 4, Figure 4.16). Furthermore, it is possible that considerable amounts of P are being leached from the modern floodplain soils, especially if these soils receive the same inputs of fertiliser as T_1 soils (Chapter 4, Figure 4.18).

Fertiliser use efficiency was found to be the lowest in the effluent treated paddocks, but, it should be recognised that this is generally not easily avoidable. Reasons for this largely pertain to financial limitations restricting the input of infrastructure required for improved storage capacities and irrigation management (Gourley 2004, Sims & Kleinman 2005). Nonetheless, the results from this study revealed that concerning P losses are highly likely for effluent treated paddocks through both surface (i.e., runoff and erosion) and subsurface (i.e., leaching and lateral translocation) pathways. However, due to the location in which the effluent treated paddocks are situated (i.e., predominantly within T_3), these losses are unlikely to be contributing concerning amounts of P to the Manning River (Appendix XVI). To confirm this, however, further investigation may be required, including a more thorough assessment of subsurface transport pathways.

The levels of Colwell P measured in surface soils from the effluent treated paddocks were far in excess of their respective recommended critical Colwell P values (Chapter 4, Figure 4.16). Thus, losses via surface runoff are highly likely to occur. In this case, however, effluent is not being applied to provide P for pasture production, but rather as a convenient means of disposal. Hence, improved pasture growth may be merely recognised as a residual benefit, although these benefits are generally not utilised to their full potential, as effluent treated pastures are typically only grazed by bulls and weaners in order to negate contamination concerns within the milk produced (Houlbrooke *et al.* 2004). This highlights the need for further research to determine the risks associated with grazing milking cows in effluent treated pastures, thereby, potentially enabling the wider adoption of effluent recycling practices to become more ethically, and economically viable (Toze 2006).

Soil profiles collected to represent effluent treated soils each had relatively high clay contents and PBI values in the subsoil depth range, but, P test results suggested that clay constituents and sorption potential (based off PBI values) are having little effect in preventing the leaching of P derived from effluent. This is because there was a lack of P accumulation (indicated by total P concentrations) in subsoils, which would otherwise (i.e., if leaching was not occurring) be high on account of the high inputs of P from effluent additions in conjunction with high clay content and PBI generally enhancing the overall soil sorption potential, thereby increasing P accumulation (Lewis *et al.* 1981).

Generally, soils with high PBI values would minimise, rather than enhance P mobility, as was highlighted previously (Chapter 5, section 5.2). However, it was found that this widely accepted assumption may not necessarily be true for P applied in the form of effluent. Moreover, the comparison of the effluent profile taken from the lower effluent treated terrace soil (T_2) as compared to the higher (also effluent treated) terrace soil (T_3) suggested that subsurface translocation of moisture, and hence, P is occurring. This is because surface soil concentrations for, pH, electrical conductivity (EC) and water extractable phosphorus (WEP) in both profiles were similar, but, all increased with depth in the lower T_2 effluent treated soil profile (Appendix I and XVI). Conversely, however, accumulation of WEP in T_2 subsurface and deep soils may be occurring due to the presence of an impermeable layer beneath the solum – a common feature of older terrace soil profiles within the study region (ASRIS 2013) .

Organic matter (OM), pH and moisture levels each can have a major influence on P dynamics (Chapter 2, section 2.5). All three of these variables are altered in the soil following the addition of effluent (Toze 2006), as effluent is typically high in OM, in liquid form, and generally has a different pH to the soil solution (Chapter 4, Table 4.3). OM and pH concentrations were both found to be negatively correlated with PBI (Chapter 4, Table 4.2), which suggests that additions of effluent may be decreasing the soils ability to sorb P. However, due to the inconsistencies in OM and pH when comparing the two effluent treated profiles (Appendix I), it is likely that fluctuating moisture levels are having the most influence on P dynamics as a result of effluent application, as moisture content was found to be relatively similar within both of these profiles.

Moisture content of soils was not quantified experimentally, but, was observed in the field. In addition, approximate moisture levels in effluent treated soils and can be inferred from other parameters, such as: clay content (i.e., expanding clays) and EC. For example, a soils susceptibility to waterlogging generally rises in conjunction with clay content, depending on the nature (e.g., expanding versus non-expanding) of clay constituents (vanLoon & Duffy 2010). Whereas, EC can be used to indicate the concentration of soluble salts, which are largely contained in the soil solution when moisture levels are high enough (Rayment & Lyons 2011); thus, EC concentrations are often used to track the movement and accumulation of moisture within the solum. Soils found to have extremely low total P concentrations (i.e., < 50) had relatively high clay contents and EC concentrations compared to samples at similar depths within the same profile with higher total P concentrations (Appendix I). Hence, it is likely that fluctuations in moisture content is minimising sorption and potentially enhancing desorption processes, reflected by the very low total P concentration.

Moisture content can have an influence on P dynamics through altering both the physical and chemical properties of a soil. Firstly, moisture levels play a major role in dictating redox conditions within the soil which may enhance the solubility of sorbed P (Chapter 2, section 2.5.8; Wu and Sansalone 2013, Yaghi and Hartikainen 2013). Secondly, P sorption (and desorption) may be influenced by wetting and drying of the soil (Chapter 2, section 2.5.8). Soils within effluent treated paddocks are constantly undergoing wetting and drying on account of the rotational nature of which the effluent is applied. This may,

therefore, be inhibiting the ability of the effluent treated soils to sorb P, and may also be enhancing desorption of previously sorbed P.

Based on the findings of this study, it is possible that effluent derived P is not being easily retained by the soil, indicating that PBI values for effluent treated soils were perhaps unrepresentative, as leaching of P is likely to be occurring. The PBI test was originally developed to estimate the soils sorption potential in response to additions of P predominantly in inorganic forms (Rayment & Lyons 2011). Therefore, it is likely that the PBI test may not be as applicable for determining P sorption for soils treated with effluent as the nature of sorption process are likely to be much different due to the chemical makeup of effluent as compared to regular fertilisers. In addition PBI may be inaccurate for these soils as they are often saturated with moisture and it has been shown that air drying soils, a requirement to conduct the PBI test, can enhance sorption potential as was previously described (Haynes 1989). These findings highlight a potential gap in the literature which may be important to address in future research.

5.4.1. Summary

For a farm of such a large scale it is not only more difficult, but, more important to maintain a stable farm-scale nutrient balance in order to minimise the risk for potential environmental losses to occur. The findings of this study suggest that Dairy A management practices have taken this into consideration and, as a result, the overall operation may be seen as relatively sustainable compared to other intensified dairies throughout Australia (Gourley *et al.* 2012). This is good news for both the farmer as well as local water catchment authorities for both financial and environmental reasons. However, the findings also revealed that certain aspects of the current nutrient management strategies could be further improved. Recommendations as to how such improvements could be made are presented in Chapter 6.

CHAPTER 6

Conclusions and Recommendations

The present study set out to extend previous research carried out on alluvial landscapes through integrating phosphorus (P) dynamics with patterns of soil profile differentiation commonly found between locations of varying height within assemblages of stepped modern floodplain and terrace units. In addition, the study was designed to assess the current farm operations in order to determine the potential for P losses to the Manning River. This chapter summarises the major findings of the study and highlights the key areas for future research. Also included, are a number of recommendations specific to the study area as to how nutrient management strategies might be improved based on the findings of the study.

6.1. Conclusions

The impetus of the study was largely driven by environmental concerns regarding the continuing intensification of the Australian dairy industry in conjunction with the rising demand, yet diminishing availability of P resources; both factors essential to the sustainability of human life. Ensuring these areas of concern do not become exacerbated strongly depends on sustainable agriculture.

The study revealed that soil profiles from varying alluvial units of increasing height above the existing channel differentiated in a pattern consistent with the supporting literature. This included a number of key trends reflecting progressive changes in soil properties from the MF through to the T_3 soil profiles, including, (1) increasing (< 2µm) clay content, (2), decreasing sand content and mean grain size, (3) increasing degree of particle size differentiation, (4) increasing kaolin, illite, goethite and gibbsite minerals, and (5) increasing Al_{ox} concentration. These trends were found to be most prominent within the subsurface (10-50 cm) and deep (> 50 cm) depth ranges of the soil profile, whereas surface soils showed relatively little variation between the four major alluvial units.

Collectively, these variations were shown to be having an influence on both soil solution movement pathways and P sorption characteristics (inferred from the PBI) of the soils; thereby, also influencing the risk of P to be lost to receiving waterbodies (Figures 5.2 and 5.3). Hence, this study confirmed that P dynamics can be expected to vary between alluvial units of alternating height in landscapes with stepped assemblages of terrace and modern floodplain units, which are common throughout eastern NSW.

There are several major implications which can be derived from the findings of the study. Firstly, routine agronomic soil testing methods may be misrepresentative of soil characteristics for a given paddock within which several alluvial units of varying height occur. Thus, inferred fertiliser requirements may be in excess of what actually necessary, at least for one of the major alluvial units. A second major implication of the study findings, therefore, is that paddocks with one or more major alluvial units of unequal height above the existing channel are likely to require variable rates of fertiliser in order to achieve optimised fertiliser use efficiency.

The knowledge gained from this research can, therefore, be applied through using it to improve farm-scale phosphorus management strategies. Due to the large number of agricultural industries operating within stepped alluvial landscapes, findings are broadly applicable throughout much of eastern NSW and perhaps other landscapes with similar geomorphological characteristics.

The assessment of the current sustainability of Dairy A revealed that there were several major areas of concern which should be addressed in order to minimise the likelihood of P losses to occur. Firstly, due to not accounting for variable soil types within paddocks consisting of two or more alluvial units of variable height, it is possible that P losses from MF soils may be of concern. Secondly, wetting and drying of effluent treated soils is potentially enhancing the mobility of P within the solum, causing it to be rapidly leached down the profile and possibly laterally translocated via preferential flow paths (perhaps facilitated by an impermeable layer below the solum). Thirdly, based on the determined critical Colwell P values, measured Colwell P levels were found to be in excess of what is required for maximum growth; thus, accumulation and subsequent losses of P via surface runoff and erosion could be minimised through better P fertiliser use efficiency.

Further research must continue to focus on improving farm sustainability in Australia, as it will prove to be crucial in maintaining the health of freshwater aquatic environment, conserving artificial P resources and providing financial benefits to farmers. This is particularly important considering the continuing intensification of the Australian dairy industry.

6.2. Recommendations

There are a number of potentially useful recommendations which can be derived from the findings of this study - some specific to the study site, whilst others are more suitable to being received by a wider audience. The recommendations have been divided into four major sections: (1) effluent treatment practices, (2) fertilisation management, (3) P loss mitigation strategies, and (4) future research directions.

6.2.1. Effluent treatment practices

The findings of the study suggest that greater efforts need to be made to ensure the sustainability of effluent treatment practices; hence, adoption of the following recommendations may be appropriate. Firstly, effluent should be applied at a lower rate to allow for maximum time between soil and effluent to enhance potential for P uptake by the soil, thereby, minimising leaching concerns. In addition, this would also prevent rapid rehydration of soils which is also unfavourable as described in the discussion. This may require altering the method of application and potentially, increasing the current effluent storage capacity. Secondly, soils regularly treated with effluent should not be allowed to dry completely, nor should they be enabled to reach field capacity; thus, it is recommended that soil moisture status is taken into account when determining which areas to treat, as opposed to convenience or time since last treated. For this to be made practical, the area which is regularly treated with effluent may sometimes need to be temporarily expanded accordingly, particularly during periods of high rainfall.

6.2.2. Fertilisation management

Management approaches to optimise fertiliser use efficiency should consider variable rate fertilisation techniques; in particular, for paddocks within which two (or more) adjacent alluvial units of unequal heights occur (e.g., modern floodplain and abandoned terrace

units). Firstly, however, it is recommended that in this situation routine agronomic testing is conducted in a way which provides results representative of each major alluvial unit rather than for the entire paddock.

Furthermore, it is also recommended that less soluble forms of P fertiliser such as rock phosphate are applied to the MF soils, for a more slowly released source of P. This may help to avoid leaching of P (potentially) without compromising production levels.

6.2.3. P loss mitigation strategies

P losses from runoff, erosion and leaching may be greatly reduced by sacrificing some of the (lower bench) modern floodplain soils in order to extend the riparian buffer zone. If this strategy was to be adopted, it is recommended that soils expected to be receiving the most surface runoff are targeted first; for example, soils at the lower end of a major backchannel. In the case of Dairy A this area relates to the SSW corner of the study area (Chapter 4, Figure 4.1).

Incorporation of more deep-rooted pasture species, such as lucerne (Medicago sativa), in modern floodplain soils is also recommended to minimise rapid P leaching. In addition, deep-rooted species may provide additional benefits through exploiting subsoil pools of leached P. For Dairy A, an effective approach may be to implement this strategy on a rotational basis in higher terraces as P leaching is likely to be occurring much more slowly, this would also reduce overall accumulation of P in subsoil making future losses increasingly less probable.

It is advised that variable rate irrigation should be implemented within paddocks where two or more distinctive alluvial units of varying height occur, as P dynamics in each alternate soil type would be expected to respond differently to additions of water. For example, leaching of P would be higher in MF soils in comparison to T_1 soils following irrigation, despite receiving the same influx of water.

6.2.4. Future research direction

Firstly, it is recommended that further research focuses on determining risks associated with feeding milking cows (i.e., milkers) effluent treated pastures. Utilising effluent as a source of P for pastures grazed for milk production would reduce inputs of P and negate the need to rely on expensive chemical P fertilisers.

Secondly, future research needs to focus on making further progress in breeding pasture plants for increased P efficiency, for example, greater utilisation of subsoil P.

Thirdly, studies must continue to develop ways in which spatial variations in alluvial soil profile characteristics can be grouped (and mapped) at a manageable scale for improved nutrient management strategies (e.g., optimised fertiliser use efficiency) to be made practically achievable.

Furthermore, study areas from previous alluvial landscape research which are utilised for agriculture (particularly dairy operations) should be investigated to see whether apparent spatial variations in soil profiles are being accounted for in farm management practices.

6.3. Summary

This study has revealed that farm P management strategies may be improved by taking into account soil variations between alluvial units of variable height above the associated channel. Consideration of these findings may help to improve the sustainability of farm practices by improving P fertiliser use efficiency.

The study also found that effluent treatment practices could be better managed in order to minimise surplus P losses and accumulation. However, these consequences are often not easily avoidable, and therefore future research needs to focus on how effluent may be better utilised as a source of P for more 'utilisable' pasture production.

Through adopting the recommendations provided in this study, the currently existing conflicting agendas of environmental authorities and agricultural sectors may be minimised as a result of improved farm sustainability, consequently satisfying the needs of the public community.

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APPENDIX I

Summary table of the total experimental data obtained and utilised in this study.

Sample ID	Depth cm	pH	EC (dS/m)	Total P mg/kg	Colwell P mg/kg	WEP mg/kg	PBI	OC % Carbon	Fe mg/kg	Al mg/kg	Sand %	Silt %	Clay %	Clay 2 µm %	D [4, 3] µm
Modern Floodplain (MF)															
Control 1	0-10	6.02	0.030	318	19	0.44	31	1.25	3946	1120	63	32	5	1	198
W1-1	0-10	5.92	0.131	537	59	3.19	49	2.47	4804	1178	53	42	5	1	152
W1-2	10-20	6.25	0.037	352	16	0.18	40	1.04	4337	1145	57	37	6	1	158
W1-3	20-30	6.50	0.028	313	14	0.08	43	0.68	4489	1296	59	34	7	2	146
W1-4	55-65	6.68	0.018	251	10	0.20	38	0.61	3926	1312	52	40	8	2	138
W1-5	105-115	6.74	0.018	268	10	0.17	43	0.65	4063	1357	55	36	9	2	151
W2-1	0-10	5.92	0.113	599	64	1.67	53	2.07	5226	1450	39	55	6	1	103
W2-2	10-20	6.28	0.042	408	18	0.13	57	1.14	5686	1658	40	50	10	2	94
W2-3	20-30	6.49	0.032	375	11	0.11	54	1.06	5854	1922	38	49	13	3	90
W2-4	130-140	6.69	0.009	253	20		29	0.14	2968	795	89	9	2	1	406
W2-5	140-150	6.97	0.013	371	25		51	0.70	5373	1552	50	34	16	10	124
W2-6	240-250										50	35	14	8	115
W2-7	390-400										68	24	8	2	197
R1-1	0-10					2.08					32	59	9	1	90
R1-2	10-20					0.88					33	57	10	2	99
R1-3	20-30					0.30					35	52	13	3	96
R1-4	70-80					0.25					72	21	7	2	267
R1-5	110-120										65	27	8	2	244
R1-6	135-145										58	27	14	8	216
R1-7	150-160										48	29	24	17	157
E1-1	0-10					15.23					46	47	7	1	134

E1-2	10-20					13.80					51	42	7	1	161
E1-3	20-30					7.38					44	46	11	2	123
E1-4	135-145					0.25					66	25	9	3	306
E1-5	180-210										72	22	7	2	311
Sample ID	**Depth cm**	**pH**	**EC (dS/m)**	**Total P mg/kg**	**Colwell P mg/kg**	**WEP mg/kg**	**PBI**	**OC % Carbon**	**Fe mg/kg**	**Al mg/kg**	**Sand %**	**Silt %**	**Clay %**	**Clay 2 µm %**	**D [4, 3] µm**
Terrace 1 Soils (T_1)															
Control 2	10	6.16	0.045	659	34	1.01	65	2.40	7320	2239	29	62	9	1	69
W3-1	0-10	5.79	0.133	1324	140	8.99	72	2.80	5089	1369	37	56	6	1	100
W3-2	10-20	6.10	0.049	625	58	1.60	56	1.55	5117	1477	38	52	10	2	100
W3-3	20-30	6.23	0.041	484	29	0.49	52	1.16	4969	1510	58	31	10	3	178
W3-4	90-100	6.72	0.019	450	59	1.22	50	0.52	4799	1393	49	27	24	18	127
W3-5	190-200										50	28	22	16	123
R2-1	0-10					5.09					37	54	9	2	103
R2-2	10-20					1.98					42	47	11	2	127
R2-3	20-30					0.49					45	43	12	3	133
R2-4	80-90										52	24	24	17	163
R2-5	200-210										55	23	22	17	158
D1-1	0-10					13.42					34	58	7	1	71
D1-2	10-20					2.72					33	58	9	2	70
D1-3	20-30					0.43					39	48	13	3	86
D1-4	85-95										52	26	22	16	122
D1-5	195-210										88	10	3	1	424
D2-1	0-10	5.94	0.061	1352	242	9.25	124	2.25	8900	2159	21	70	9	1	47
D2-2	10-20	6.06	0.038	906	117	2.73	102	1.68	8548	2168	41	48	12	3	131
D2-3	20-30	6.28	0.030	705	62	1.29	91	1.39	7935	2215	25	61	14	3	58
D2-4	85-95	6.37	0.036	528	49	0.50	77	0.97	7458	2191	27	53	19	8	55
D2-5	190-205										37	37	26	19	73
BJ2-1	0-10					4.82					18	70	12	2	54

BJ2-2	140-150					3.29					27	47	26	17	45
BJ3-1	0-10					17.21					32	58	10	2	81
BJ3-2	10-20					5.59					34	53	14	3	84
BJ3-3	20-30					1.20					36	50	15	3	83
BJ3-4	90-100					1.67					38	37	26	19	79
BJ3-5	190-205										36	36	27	20	68
Sample ID	**Depth cm**	**pH**	**EC (dS/m)**	**Total P mg/kg**	**Colwell P mg/kg**	**WEP mg/kg**	**PBI**	**OC % Carbon**	**Fe mg/kg**	**Al mg/kg**	**Sand %**	**Silt %**	**Clay %**	**Clay 2 µm %**	**D [4, 3] µm**
Terrace 2 Soils (T_2)															
BJ1-1	0-10					0.17					21	66	13	3	56
BJ1-2	30-38					0.15					23	58	19	11	62
BJ1-3	40-48					3.20					18	52	30	22	47
BJ1-4	48-53					5.54					14	50	36	26	55
BJ1-5	65-75					6.39					7	52	41	29	22
BJ1-6	80-90					7.79					14	56	31	21	30
BJ1-7	90-96					9.65					19	49	32	23	44
BJ1-8	96-100					13.04					25	48	27	19	77
BJ1-9	140-150					15.02					20	66	15	4	55
AJ2-1	0-10	5.90	0.072	358	30	0.54	148	1.39	7725	2271	19	70	11	3	49
AJ2-2	30-40	5.71	0.073	136	3	0.20	179	0.39	4232	1730	17	67	16	6	52
AJ2-3	40-50	5.51	0.095	112	3	0.13	282	0.42	5195	2469	11	68	21	8	31
AJ2-4	60-70	5.43	0.105	153	4	0.17	336	0.33	6040	2460	11	58	32	21	25
AJ2-5	70-80	5.55	0.092	139	4	0.17	191	0.26	4743	1885	14	62	24	14	33
AJ2-6	80-88	5.54	0.085	116	4	0.30	169	0.20	3515	1615	12	60	29	18	31
AJ2-7	110-120					0.72					13	52	35	25	37
AJ2-8	120-130	5.45	0.120	*50*	2	3.72	169	0.31	4101	1687	0	75	25	8	15
AJ2-9	130-140					9.25					9	51	40	30	23
AJ2-10	140-150					1.70					15	47	38	30	40
AJ3-1	0-10					0.26					24	66	10	2	69

AJ3-2	10-20					0.33					16	66	18	8	41
AJ3-3	30-40					0.25					19	66	15	5	53
AJ3-4	50-60					0.88					8	55	37	25	23
AJ3-5	80-90					1.92					13	44	44	34	26
EC1-1	0-5	6.00	0.262	970	383	31.92	118	5.65	2474	1109	34	60	6	1	81
EC1-2	5-10					33.39					37	56	7	1	96
EC1-3	10-15					21.77					35	57	8	1	87
EC1-4	15-20					3.38					24	64	12	3	55
EC1-5	20-25	6.52	0.047	248	25	0.36	82	1.06	4376	1404	31	57	12	4	87
EC1-6	25-30					0.20					23	58	19	5	77
EC1-7	30-35					0.22					16	66	18	5	49
EC1-8	35-45	6.89	0.064	142	26	0.99	263	0.59	4634	3351	16	60	24	13	37
EC1-9	45-55					2.22					41	41	18	12	146
EC1-10	55-65	7.61	0.092	168	14	3.36	203	0.31	5336	2136	26	53	21	10	78
EC1-11	65-75					14.52					22	42	36	28	66
EC1-12	75-90	7.17	0.467	*50*	24	18.21	137	0.29	2114	1658	13	45	42	34	36
Sample ID	**Depth cm**	**pH**	**EC (dS/m)**	**Total P mg/kg**	**Colwell P mg/kg**	**WEP mg/kg**	**PBI**	**OC % Carbon**	**Fe mg/kg**	**Al mg/kg**	**Sand %**	**Silt %**	**Clay %**	**Clay 2 µm %**	**D [4, 3] µm**
Terrace 3 Soils (T_3)															
AJ-1-1	0-10	6.60	0.218	1083	453	52.74	125	3.07	5096	1510	28	63	9	2	76
AJ-1-2	20-27	6.30	0.104	369	37	0.88	113	1.31	3379	1735	32	54	14	4	104
AJ-1-3	27-33	5.60	0.137	226	28	1.26	211	0.90	2866	2349	28	46	26	13	71
AJ-1-4	33-40	5.22	0.123	177	15	0.74	424	0.77	2641	3084	11	40	48	34	32
AJ-1-5	40-50	5.34	0.099	205	4	2.20	565	0.57	4039	3895	36	45	19	5	95
AJ-1-6	50-60	5.40	0.110	59	3	1.76	484	0.40	1957	9456	14	52	34	20	42
AJ-1-7	60-67	5.35	0.109	*50*	2	1.34	414	0.28	2427	3277	20	41	39	28	170
AJ-1-8	80-87	5.35	0.091	90	5	5.71	203	0.22	2530	1992	19	40	41	31	45
AJ-1-9	90-100	5.34	0.100	71	10	3.98	148	0.24	1321	1560	32	40	28	20	92
AJ-1-10	100-107					1.98					39	49	12	3	104

AJ-1-11	107-113					1.09					45	47	9	2	123
AJ-1-12	120-130	5.37	0.091	367	3	0.99	102	0.16	2416	1550	55	38	7	2	173
Sample ID	**Depth cm**	**pH**	**EC (dS/m)**	**Total P mg/kg**	**Colwell P mg/kg**	**WEP mg/kg**	**PBI**	**OC % Carbon**	**Fe mg/kg**	**Al mg/kg**	**Sand %**	**Silt %**	**Clay %**	**Clay 2 µm %**	**D [4, 3] µm**
Control															
Control 3-1	0-10	5.71	0.032	698	53	0.74	82	2.27	12018	2135	24	65	12	3	53
Control 3-2	10-20	5.93	0.019	600	31	0.32	80	1.80	12105	2208	26	60	14	4	64
Control 3-3	20-30	6.07	0.016	567	23	0.29	95	1.64	12271	2442	29	53	18	8	94

Values in italics represent results which were below the limit of detection (LOD), where the value reported is the limit of detection.
*D [4, 3] = volume weighted mean grain size (µm).

APPENDIX II

Mineralogy results (raw data)

Alluvial Unit	Sample	Depth (cm)	Quartz	Albite	Orthoclase	Labradorite	Microcline	Muscovite	*Illite	Kaolin	Goethite	Gibbsite
	Control 1	0-10	67.5	17.2	9.2	2.1	3	0.1	0.4	0.7	-	-
	W1-2	10-20	64.6	20.9	5.7	2.8	0.4	3.4	0.5	0.6	-	0.7
	W 1-3	20-30	63.4	19.8	4.2	4.1	2	2.5	1.1	1.4	-	0.9
	W1-4	55-65	64.5	21.1	3.4	4.1	-	3.1	0.5	1.8	-	0.4
MF	W 2-2	10-20	55.4	30.3	4	2	1.7	3.5	0.5	2	-	0.2
	W 2-4	130-140	68.3	22.5	3.8	0.6	0.2	0.8	0.4	1.3	-	0.9
	W2-5	140-150	61.1	19.3	7.4	4.6	-	1.6	2	1.4	-	0.4
	R 1-2	10-20	59.5	25.4	-	7.6	-	5.5	0.4	1.5	-	-
	E1 -2	10-20	76	23.3	-	-	-	-	0.3	0.3	-	-
	Control 2	0-10	52.6	25.3	3.9	9.5	0.2	5.5	1.1	1.4	-	0.5
	W 3-2	10-20	69.8	22.7	4.8	-	-	1.3	0.7	0.5	-	-
	W 3-4	90-100	59.2	24.6	5	3	-	5.4	0.8	1.3	-	0.7
	W 3-5	190-200	54.7	21.7	5.8	7.4	4.5	4.8	0.4	1.2	-	0.2
	R 2-2	10-20	68.3	25.9	3	-	-	-	0.4	0.9	-	0.5
T1	D 1-2	10-20	50.8	22.8	-	17.7	-	8	0.3	0.4	-	-
	D 1-4	85-95	51.1	31.3	9.2	5.7	-	1.8	0.4	0.4	-	0.1
	D 1-5	195-210	72.5	18.1	3.8	0.4	0.7	3.3	0.3	0.3	-	0.1
	D 2-2	10-20	38.6	24.3	7.4	14	6	4	2.6	2.1	0.2	0.8
	BJ 3 -2	10-20	55.7	18.4	2.7	10.6	4.8	-	4.3	1.5	-	0.2
	BJ 3-4	90-100	47.8	26.8	7.5	4.3	6.4	4.1	0.5	1.6	-	0.8
	BJ 3-5	190-205	58.6	28	-	0.9	1.1	4.9	3.6	1.8	0.2	0.7
	EC 1-3	10-20	77.4	17.7	1.4	1.6	-	-	1.1	0.6	-	0.5
	EC 1-5	20-25	75.9	19.8	1.2	-	-	0.7	0.4	0.4	-	0.9
	EC 1-8	35-45	79.5	11.1	-	-	-	-	3.5	2.8	1.9	1.2

	EC 1-12	75-90	64.15	14.6	1.2	-	-	3.95	6.2	5.9	1.8	1.95
	AJ 2-1	0-10	86.2	12.1	-	-	-	-	0.5	0.4	-	0.6
	AJ-2-2	30-40	84	5.2	5.1	-	-	-	0.9	2.1	2.0	0.7
	AJ 2-5	70-80	82.4	7.7	-	-	-	2.7	2.9	2.4	0.7	1.1
T2	AJ 2-10	140-150	51.8	28.8	-	-	-	4.3	6	6.3	0.4	2.3
	AJ-3-1	0-10	81.6	3	6.1	-	2.3	-	-	1.9	0.6	1.1
	AJ-3-2	10-20	83.1	11	0.8	-	-	-	1.3	1.3	1.2	1.2
	AJ 3-5	80-90	65.4	11.2	-	-	-	7.9	4.4	6.7	2.2	2.2
	BJ-1-1	0-10	80.4	10.3	2.4	-	0.9	0.6	1.4	1.2	1.4	0.9
	BJ-1-2	30-38	81.3	9.8	1.8	-	-	-	1.8	1.4	0.9	1
	BJ-1-5	65-75	56	7.4	-	-	-	6.5	11.9	11.7	3.2	2.7
	BJ-1-7	90-96	69.1	19.7	1.1	-		4.6	1.2	3.8	-	0.4
	AJ 1-2	20-27	84.1	1.1	6.8	0.2	2	1.2	0.4	1.6	-	-
	AJ-1-3	27-33	88.8	1	-	-	-	1	0.4	4.3	-	-
	AJ 1-4	33-40	22.5	38.6	2.5	6.6	-	10.4	6.4	11.7	-	1.3
	AJ-1-6	50-60	34.3	8.5	-	-	-	14.3	16.6	19.6	2.5	3.6
T3	AJ-1-7	60-67	34.7	12.3	-	-	-	14.3	15.3	16.3	2.4	3.1
	AJ-1-9	90-100	23.6	23.8	1.9	7.7	1.9	12.4	11.1	14.1	0.8	2.1
	AJ-1-10	100-107	17.5	50.6	0.7	-	-	13.9	4.5	9.9	0.9	1.9
	AJ 1-12	120-130	21.4	41.6	0.6	1.8	-	12.5	4.6	10.2	3.3	2.2
Control Terrace	Control 3-1	0-10	60.2	27.5	2.4	-	-	3.6	2.6	1.9	-	0.6

*Inclusive of illite and mixed layer illite

Mineralogy results (summary)

Alluvial Unit	Sample	Depth (cm)	*Mineralogy**						
			Quartz	**Feldspars**[†]	**Muscovite**	**Illite**[‡]	**Kaolin**	**Goethite**	**Gibbsite**
MF	Control 1	0-10	high	moderate	trace	trace	trace	-	-
	W1-2	10-20	high	moderate	trace	trace	trace	-	trace
	W 1-3	20-30	high	moderate	trace	trace	trace	-	trace
	W1-4	55-65	high	moderate	trace	trace	trace	-	trace
	W 2-2	10-20	high	moderate	trace	trace	trace	-	trace
	W 2-4	130-140	high	moderate	trace	trace	trace	-	trace
	W2-5	140-150	high	moderate	trace	trace	trace	-	trace
	R 1-2	10-20	high	moderate	low	low	trace	-	-
	E1 -2	10-20	high	moderate	-	trace	trace	-	-
T_1	Control 2	0-10	high	moderate	low	low	trace	-	trace
	W 3-2	10-20	high	moderate	trace	trace	trace	-	-
	W 3-4	90-100	high	moderate	low	low	trace	-	trace
	W 3-5	190-200	high	moderate	trace	low	trace	-	trace
	R 2-2	10-20	high	moderate	-	trace	trace	-	trace
	D 1-2	10-20	high	high	low	low	trace	-	-
	D 1-4	85-95	high	high	trace	trace	trace	-	trace
	D 1-5	195-210	high	moderate	trace	trace	trace	-	trace
	D 2-2	10-20	moderate	high	trace	low	trace	trace	trace
	BJ 3 -2	10-20	high	moderate	-	trace	trace	-	trace
	BJ 3-4	90-100	high	high	trace	trace	trace	-	trace
	BJ 3-5	190-205	high	moderate	trace	low	trace	trace	trace
T_2	EC 1-3	10-15	high	moderate	-	trace	trace	-	trace
	EC 1-5	20-25	high	moderate	trace	trace	trace	-	trace
	EC 1-8	35-45	high	low	-	trace	trace	trace	trace

	EC 1-12	75-90	high	low	trace	low	low	trace	trace
	AJ 2-1	0-10	high	low	-	trace	trace	-	trace
	AJ-2-2	30-40	high	low	low	-	-	trace	
	AJ 2-5	70-80	high	low	trace	low	trace	trace	trace
	AJ 2-10	140-150	high	moderate	trace	low	low	trace	trace
	AJ-3-1	0-10	high	low	-	-	trace	trace	trace
	AJ-3-2	10-20	high	low	-	trace	trace	trace	trace
	AJ 3-5	80-90	high	low	low	low	low	trace	trace
	BJ-1-1	0-10	high	low	trace	trace	trace	trace	trace
	BJ-1-2	30-38	high	low	-	trace	trace	trace	trace
	BJ-1-5	65-75	high	low	-	trace	trace	trace	trace
	BJ-1-7	90-96	high	moderate	low	trace	trace	-	trace
	AJ 1-2	20-27	high	low	low	trace	trace	trace	-
T_3	AJ-1-3	27-33	high	trace	trace	trace	trace	-	-
	AJ 1-4	33-40	moderate	high	low	low	low	-	trace
	AJ-1-6	50-60	moderate	low	low	low	moderate	trace	trace
	AJ-1-7	60-67	moderate	moderate	low	low	low	trace	trace
	AJ-1-9	90-100	moderate	moderate	low	low	low	trace	trace
	AJ-1-10	100-107	low	high	low	low	low	trace	trace
	AJ 1-12	120-130	moderate	high	low	low	low	trace	trace
Control Terrace	Control 3-1	0-10	high	moderate	trace	low	trace	-	trace

*Terms used to describe mineralogy represent the following percentage ranges (determined to the nearest whole value): high = >40; moderate = 20-40; low = 5-20; and trace = 0-5

†Inclusive of: albite, orthoclase, labrodorite and microcline; ‡ Inclusive of illite and mixed layer illite

APPENDIX III

Analysis of variance between key soil properties among the four major alluvial units using the Kruskal-Wallis test

Depth Range	Parameter	All locations	MF vs. T_1	MF vs. T_2	MF vs. T_3	T_1 vs. T_2	T_1 vs. T_3	T_2 vs. T_3
Surface (0-10 cm)	Sand	p = 0.151	p = 0.068	p = 0.050	p = 0.157	p = 0.711	p = 0.614	p = 0.770
	Clay	p = 0.200	p = 0.109	p = 0.079	* p = 0.046	p = 0.375	p = 0.386	p = 1 .000
	**D [4, 3]	p = 0.082	* p = 0.033	* p = 0.027	p = 0.157	p = 0.584	p = 1.000	p = 0.770
	Al_{0x}	p = 0.816	p = 0.439	p = 1.000	p = 0.221	p = 1.000	p = 1.000	p = 1.002
	Fe_{ox}	p = 0.887	* p = 0.041	p = 1.000	p = 1.000	p = 0.439	p = 1.000	p = 1.000
	PBI	p = 0.156	p = 0.121	p = 0.121	p = 0.221	p = 0.439	p = 0.221	p = 1.00
Subsurface (10-50 cm)	Sand	* p = 0.000	p = 0.328	* p = 0.000	* p = 0.017	* p = 0.000	* p = 0.040	p = 0.247
	Clay	* p = 0.000	* p = 0.039	* p = 0.002	* p = 0.005	* p = 0.002	* p = 0.002	p = 0.501
	**D [4, 3]	* p = 0.001	p = 0.248	* p = 0.000	p = 0.089	* p = 0.002	p = 0 .322	p = 0.225
	Al_{0x}	p = 0.112	p = 0.248	p = 0.149	* p = 0..043	p = 0.564	p = 0.083	p = 0.386
	Fe_{ox}	* p = 0.015	p = 0.248	p = 0.386	* p = 0.021	p = 0.083	* p = 0.021	* p = 0.021
	PBI	* p = 0.011	p = 0.149	* p = 0.021	* p = 0.021	: p = 0.081	* p = 0.021	: p = 0.386
Deep (> 50 cm)	Sand	* p = 0.000	* p = 0.012	* p = 0.000	* p = 0.002	* p = 0.000	p = 0.123	* p = 0.08
	Clay	* p = 0.000	* p = 0.003	*: p = 0.000	p = 0.059	* p = 0.024	p = 0.495	p = 0.203
	**D [4, 3]	* p = 0.000	* p = 0.014	* p = 0.000	* p = 0.009	* p = 0.000	p = 0.892	* p = 0.004
	Al_{0x}	p = 0.052	p = 0.165	* p = 0.011	* p = 0.027	p = 0.739	p = 0.439	p = 0.855
	Fe_{ox}	* p = 0.024	p = 0.165	p = 0.670	* p = 0.014	p = 0.182	p = 0.053	* p = 0.028
	PBI	* p = 0.010	p = 0.165	* p = 0.010	* p = 0.014	* p = 0.044	p = 0.053	: p = 0.647
Cumulative	Sand	* p = 0.000	* p = 0.001	* p = 0.000	* p = 0.000	* p = 0.000	* p = 0.037	* p = 0.008
	Clay	* p = 0.000	* p = 0.016	* p = 0.000	* p = 0.001	* p = 0.003	p = 0.072	p = 0.694
	**D [4, 3]	* p = 0.000	* p = 0.000	* p = 0.000	* p = 0.004	* p = 0.000	p = 0.648	* p = 0.003
	B/A ratio	p = 0.422	p = 0.238	p = 0.180	p = 0.277	p = 0.516	p = 0.614	p = 0.709
	Al_{0x}	* p = 0.004	* p = 0.021	* p = 0.008	* p = 0.002	p = 0.440	= 0.083	p = 0.356
	Fe_{ox}	* p = 0.000	* p = 0.041	p = 0.692	* p = 0.002	* p = 0.017	* p = 0.001	* p = 0.010
	PBI	* p = 0.000	* p = 0.004	* p = 0.000	* p = 0.000	* p = 0.001	* p = 0.001	p = 0.391

*p is significant at the 0.05 level; ** D [4, 3] = volume weighted mean grain size (μm).

APPENDIX IV

Analysis of variance (Kruskal-Wallis) between depth ranges

Alluvial Unit	Parameter	*All depth ranges	Surface vs. Subsurface	Subsurface vs. Deep	Surface vs. Deep
	Sand	* p = 0.007	p = 0.734	* p = 0.009	* p = 0.015
MF	Clay	* p = 0.010	* p = 0.026	p = 0.185	* p = 0.000
	**D [4, 3]	* = 0.011	p = 1.000	* p = 0.009	* p = 0.029
	Al_{ox}	p = 0.229	p = 0.064	p = 0.248	= 0.335
	Fe_{ox}	p = 0.424	p = 0.643	p 0.248	p =-0.355
	PBI	p = 0.472	p = 0.355	p = 0.355	p = 0.439
	Sand	p = 0.600	= 0.550	p = 0.324	p = 0.801
	Clay	* p = 0.000	* p = 0.003	* p = 0.001	* p = 0.004
T_1	**D [4, 3]	p = 0.251	p = 0.104	p = 0.778	p = 0.191
	Al_{ox}	p = 0.077	p = 0.165	p = 0.064	p = 0.121
	Fe_{ox}	p = 0.717	p = 0.100	= 0.100	p = 0.121
	PBI	p = 0.472	p = 0.355	p = 0.355	p = 0.439
	Sand	* p = 0.012	p = 0.090	* p = 0.031	* p = 0.019
T_2	Clay	* p = 0.000	* p = 0.008	* p = 0.000	* p = 0.001
	**D [4, 3]	* p = 0.019	p = 0.140	* p = 0.047	* p = 0.021
	Al_{ox}	p = 0.532	p = 0.643	p = 0.394	p = 0.317
	Fe_{ox}	p = 0.344	p = 0.643	p = 0.201	p = 0.114
	PBI	* = 0.013	p = 0.064	* p = 0.010	* p = 0.002
	Sand	p = 0.807	p = 0.717	p = 0.507	p = 0.827
T_3	Clay	p = 0.385	p = 0.157	p = 0.704	p = 0.261
	**D [4, 3]	p = 0.572	p = 1.000	p = 0.345	p = 0.513
	Al_{ox}	p = 0.266	p = 0.157	p = 0.624	p = 0.143
	Fe_{ox}	* p = 0.025	p = 0.157	* p = 0.014	p = 0.143
	PBI	p = 0.558	p = 0.480	p = 0.460	p = 0.380

*p is significant at the 0.05 level; **D [4, 3] = volume weighted mean grain size (μm)
Profile depth range values: surface = 0-10 cm, subsurface = 0-50 cm and deep = >50 cm.

APPENDIX V

Analysis of correlations with location

	PBI	Sand	Silt	Clay 2um	D [3, 4]	*PSD	Al_{ox}	Fe_{ox}
Rs	0.835	-0.668	0.398	0.489	-0.571	0.461	0.562	-0.515
P	0.000	0.000	0.000	0.000	0.000	0.084	0.000	0.001
n	40	99	99	99	99	15	40	40

*Particle size differentiation (PSD)

Location is a variable used to represent the sequence of stepped alluvial units; e.g., four locations: MF, T_1, T_2 and T_3.

APPENDIX VI

Analysis of correlations between soil properties and depth determined using the Spearman Rank Order Correlation test

		Correlation coefficient r_s	Sig. 2-tailed	N
MF	Sand	0.640	0.001	24
	Clay	0.585	0.003	24
	*D [4, 3]	0.598	0.002	24
	Al_{ox}	0.545	0.103	10
	Fe_{ox}	-0.493	0.148	10
	PBI	-0.508	0.134	10
T_1	Sand	0.098	0.628	27
	Clay	0.800	0.000	27
	*D [4, 3]	0.208	0.297	27
	Al_{ox}	0.849	0.008	8
	Fe_{ox}	-0.386	0.345	8
	PBI	-0.463	0.248	8
T_2	Sand	-0.500	0.002	36
	Clay	0.787	0.000	36
	*D [4, 3]	-0.473	0.004	36
	Al_{ox}	-0.313	0.322	12
	Fe_{ox}	-0.099	0.759	12
	PBI	0.673	0.016	12
T_3	Sand	0.196	0.541	12
	Clay	0.113	0.727	12
	*D [4, 3]	0.315	0.319	12
	Al_{ox}	0.181	0.617	10
	Fe_{ox}	-0.905	0.000	10
	PBI	-0.007	0.985	10

**D [4, 3] = volume weighted mean grain size (µm)

APPENDIX VII.

Particle size differentiation calculations

Alluvial Unit	Profile ID	A horizon clay (< 2 µm) minimum	B horizon Clay (< 2µm) maximum	B/A ratio
	W1	1	2	2
MF	W2	1	10	10
	R1	1	17	17
	E1	1	3	3
	W3	1	18	18
T_1	R2	2	17	9
	D1	1	16	16
	D2	1	19	19
	BJ2	2	17	9
	BJ3	2	20	10
	BJ1	3	29	10
T_2	AJ2	3	30	10
	AJ3	2	34	17
	EC1	1	34	34
T_3	AJ1	2	34	17

A horizon = surface soils, B horizon includes both subsurface (10-50 cm) and deep (> 50 cm) soils.

APPENDIX VIII.

Calculated means for chemical parameters determined by commercial analysis

Alluvial Unit	N	Depth fraction	OC (% carbon)	pH	EC (dS/m)	Total P (mg/kg)	Colwell P (mg/kg)
	2	Surface	2.27	5.92	0.122	568	62
MF	4	Subsoil	0.98	6.38	0.035	362	15
	4	Deep	0.53	6.77	0.015	286	16
	2	Surface	2.53	5.87	0.097	1338	191
T_1	4	Subsoil	1.45	6.17	0.040	680	67
	2	Deep	0.75	6.55	0.028	489	54
	1	Surface	1.39	5.90	0.072	358	30
T_2 (control)	2	Subsoil	0.41	5.61	0.084	124	3
	4	Deep	0.28	5.49	0.101	115	4
	1	Surface	5.65	6.00	0.262	970	383
T_2 (effluent)	2	Subsoil	0.83	6.71	0.056	195	26
	2	Deep	0.30	7.39	0.280	109	19
	1	Surface	3.07	6.60	0.218	1083	453
T_3 (effluent)	4	Subsoil	0.89	5.62	0.116	244	21
	5	Deep	0.26	5.36	0.100	127	5

Profile depth fraction values are as follows: surface = 0-10 cm; subsurface = 10-50 cm; deep = >50 cm.

APPENDIX IX

Calculated mean oxalate extractable iron (Fe_{ox}) and aluminium (Al_{ox}) concentration values.

Alluvial Unit	N	Fe_{ox} (mg/kg)	Al_{ox} (mg/kg)
	2	5015	1314
MF	4	5092	1505
	4	4083	1254
	2	6995	1764
T_1	4	6642	1843
	2	6129	1792
	2	5100	1690
T_2	4	4609	2239
	6	4162	1904
	1	5096	1510
T_3	4	3231	2766
	5	2130	3567

APPENDIX X

Calculated means for water extractable phosphorus (WEP) concentrations

Alluvial Unit	N	Depth fraction	WEP (mg/kg)
MF	4	Surface	5.54
	8	Subsurface	2.86
	4	Deep	0.22
T_1	6	Surface	9.80
	10	Subsurface	1.85
	4	Deep	1.67
T_2 (control)	3	Surface	0.33
	6	Subsurface	0.71
	15	Deep	5.08
T_2 (effluent)	2	Surface	32.65
	6	Subsurface	4.49
	4	Deep	9.58
T_3 (effluent)	1	Surface	52.74
	4	Subsurface	1.27
	7	Deep	2.41

Profile depth range values are as follows: surface = 0-10 cm; subsurface = 10-50 cm; deep = >50 cm.

APPENDIX XI

Calculated mean PBI values for each terrace, including all three depth ranges

Alluvial Unit	N	Depth fraction	PBI
MF	2	Surface	51
	4	Subsoil	49
	4	Deep	40
T_1	2	Surface	98
	4	Subsoil	75
	2	Deep	64
T_2	2	Surface	133
	4	Subsoil	202
	6	Deep	193
T_3	1	Surface	125
	4	Subsoil	328
	5	Deep	270

Profile depth range values are as follows: surface = 0-10 cm; subsurface = 10-50 cm; deep = >50 cm.

APPENDIX XII

Water extractable phosphorus (WEP) SPSS Spearmans rank-order correlation outputs

Correlation between WEP and EC for T_2 (effluent) soils

			WEP	EC
Spearman's rho	WEP	Correlation Coefficient	1.000	.900*
		Sig. (2-tailed)	.	.037
		N	12	5
	EC	Correlation Coefficient	.900*	1.000
		Sig. (2-tailed)	.037	.
		N	5	5

* Correlation is significant at the 0.05 level (2-tailed).

Correlation between WEP and EC for T_2 (control) soils

EC			WEP	EC
Spearman's rho	WEP	Correlation Coefficient	1.000	-.144
		Sig. (2-tailed)	.	.758
		N	10	7
	EC	Correlation Coefficient	-.144	1.000
		Sig. (2-tailed)	.758	.
		N	7	7

* Correlation is significant at the 0.05 level (2-tailed)

APPENDIX XIII

Critical Colwell P values calculated for commercially analysed samples based on PBI values, including comparison to measured Colwell P

Alluvial Unit	Sample ID	Depth fraction	Measured PBI	PBI Category	CV range (mg/kg)	CV mid-point (mg/kg)	Measured Colwell P (mg/kg)
	W1-1	Surface	49	Very low	27-31	29	59
	W1-2	Subsurface	40	Very low	27-31	29	16
	W1-3	Subsurface	43	Very low	27-31	29	14
	W1-4	Deep	38	Very low	27-31	29	10
MF	W1-5	Deep	43	Very low	27-31	29	10
	W2-1	Surface	53	Very low	27-31	29	64
	W2-2	Subsurface	57	Very low	27-31	29	18
	W2-3	Subsurface	54	Very low	27-31	29	11
	W2-4	Deep	29	Very very low	24-27	26	20
	W2-5	Deep	51	Very low	27-31	29	25
	W3-1	Surface	72	Low	31-36	34	140
	W3-2	Subsurface	56	Very low	27-31	29	58
	W3-3	Subsurface	52	Very low	27-31	29	29
T_1	W3-4	Deep	50	Very low	27-31	29	59
	D2-1	Surface	124	Low	31-36	34	242
	D2-2	Subsurface	102	Low	31-36	34	117
	D2-3	Subsurface	91	Low	31-36	34	62
	D2-4	Deep	77	Low	31-36	34	49
	AJ2-1	Surface	148	Moderate	36-44	40	30
	AJ2-2	Subsurface	179	Moderate	36-44	40	3
	AJ2-3	Subsurface	282	High	44-64	55	3
T_2 (control)	AJ2-4	Deep	336	High	44-64	55	4
	AJ2-5	Deep	191	Moderate	36-44	40	4
	AJ2-6	Deep	169	Moderate	36-44	40	4

	AJ2-8	Deep	169	Moderate	36-44	40	2
	EC1-1	Surface	118	Low	31-36	34	383
	EC1-5	Subsurface	82	Low	31-36	34	25
T_3 (effluent)	EC1-8	Subsurface	263	Moderate	36-44	40	26
	EC1-10	Deep	203	Moderate	36-44	40	14
	EC1-12	Deep	137	Low	31-36	34	24
	AJ-1-1	Surface	125	Low	31-36	34	453
	AJ-1-2	Subsurface	113	Low	31-36	34	37
	AJ-1-3	Subsurface	211	Moderate	36-44	40	28
	AJ-1-4	Subsurface	424	High	44-64	55	15
T_3 (effluent)	AJ-1-5	Subsurface	565	High	44-64	55	4
	AJ-1-6	Deep	484	High	44-64	55	3
	AJ-1-7	Deep	414	High	44-64	55	2
	AJ-1-8	Deep	203	Moderate	36-44	40	5
	AJ-1-9	Deep	148	Moderate	36-44	40	10
	AJ-1-12	Deep	102	Low	31-36	34	3

Terms used to indicate PBI categories represent the following PBI value ranges: extremely low = < 15; very very low = 15-35; very low = 36-70; low = 71-140; moderate = 141-280; high = 280-840; very high = >840 (taken from Gourley *et al.* 2007a).
Profile depth range values are as follows: surface = 0-10 cm; subsurface = 10-50 cm; deep = >50 cm.

APPENDIX XIV

Miscellaneous SPSS outputs

Correlation between total P and depth

			Total P	Depth Fraction
Spearman's rho	Total P	Correlation Coefficient	1.000	-.624**
		Sig. (2-tailed)	.	**.000
		N	40	40
	Depth Range	Correlation Coefficient	-.624**	1.000
		Sig. (2-tailed)	**.000	.
		N	40	80

**. Correlation is significant at the 0.01 level (2-tailed).

Includes all commercially analysed samples (excluding controls).

APPENDIX XV

National Australian dairy farm data from the Australian Government Department of Agriculture (ABARES 2015).

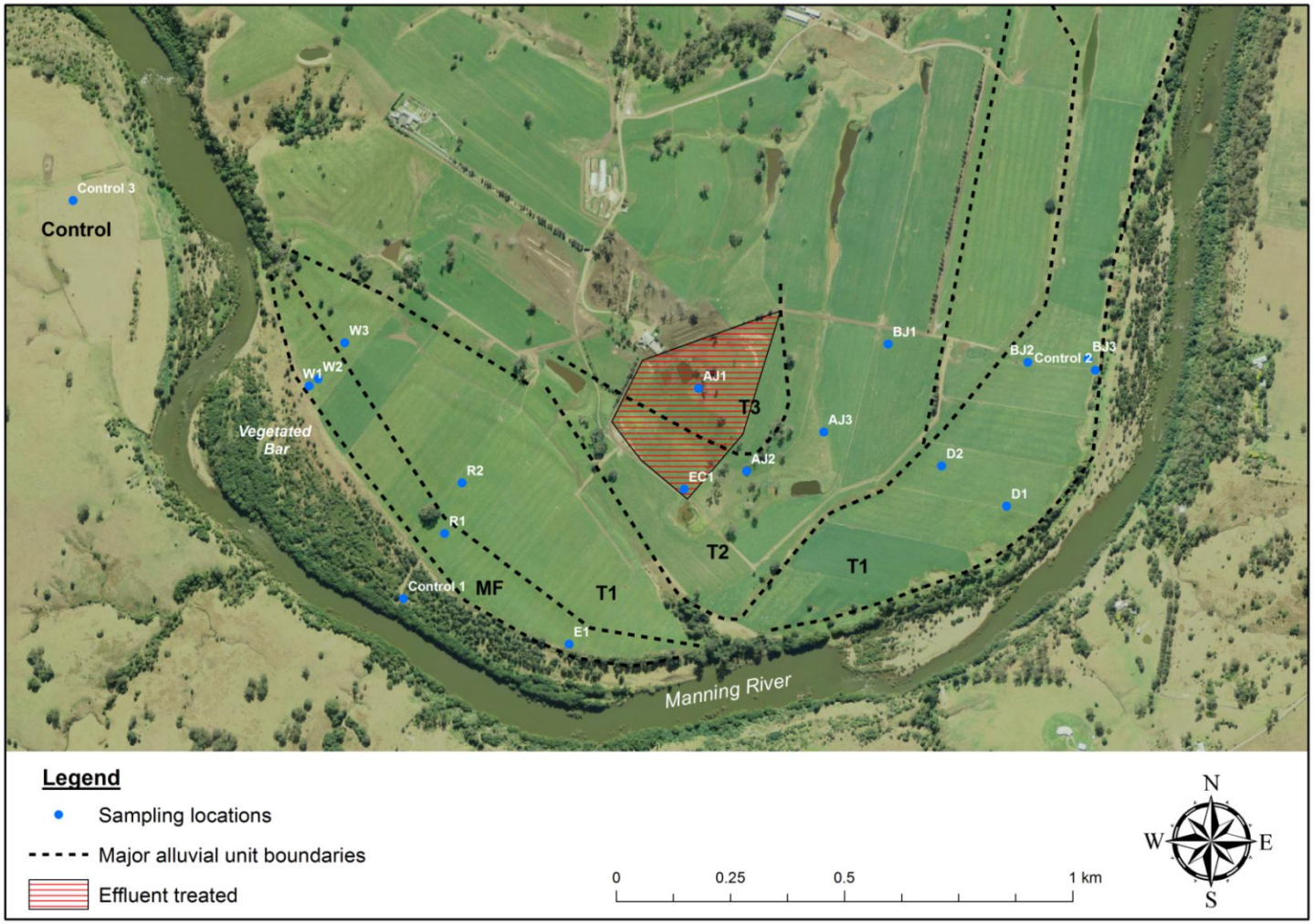
Year	Dairy cows at 30 June (no.)	RSE	Dairy cattle at June 30 (no.)	RSE	Area operated at June 30 (ha)	RSE	Milk produced (L)	RSE	No. Farms
1990	113	2	168	2	161	4	426869	3	14453
1991	116	2	176	3	181	5	445330	3	13851
1992	120	3	182	3	178	4	472254	4	13592
1993	122	2	187	2	182	4	491054	3	13607
1994	126	2	195	2	189	4	541258	2	14059
1995	145	2	225	2	219	5	594412	3	13854
1996	148	3	227	3	214	5	617813	4	13674
1997	159	3	246	3	208	5	649488	4	13376
1998	161	3	250	2	215	4	684012	3	13246
1999	165	3	258	3	216	5	769041	4	12781
2000	165	3	256	3	217	4	805177	4	12960
2001	170	3	263	3	217	6	811252	3	12602
2002	197	3	310	2	257	5	994405	4	10995
2003	192	3	303	3	253	5	921897	4	10707
2004	-	-	301	4	236	5	962805	4	10178
2005	-	-	294	3	226	5	973442	4	10132
2006	209	3	322	3	247	6	1081595	3	9361
2007	200	3	310	3	244	6	1044473	3	9081
2008	215	4	334	4	252	5	1136925	4	8106
2009	218	4	342	3	263	5	1245084	4	7500
2010	216	5	340	4	249	5	1204824	4	7514
2011	219	4	347	4	236	5	1205639	4	7501
2012	232	5	374	5	254	5	1316395	5	7233
2013	239	7	387	7	289	6	1306076	8	7087
2014	243	6	397	7	286	8	1319478	5	6993

Item Descriptions

1. Dairy cows at 30 June (no.)
    - Number of dairy cows on hand at 30 June.
2. Dairy cattle at June 30 (no.)
    - Number of dairy cattle on hand 30 June.
3. Area operated at 30 June (ha)
    - Includes all land operated by the farm business at the end of June whether owned or rented by the business. Land sharefarmed on another farm is excluded.
4. Manufacturing milk (L)
    - The total amount of milk produced for use in manufactured dairy products including butter, cheese and cream.
5. Milk produced (L)
    - Total volume of milk sold during the survey year.
6. RSE (%)
    - Relative standard error

APPENDIX XVI

Map of study area (Dairy A) showing labelled sampling sites as well as effluent treated areas



APPENDIX XVII

Detailed overview of agronomic and environmental phosphorus tests

Agronomic tests

Agronomic P tests are predominantly used to estimate fertiliser requirements (Wang *et al.* 2015). Thus, traditional methods of agronomic sample collection generally involve sampling the top 0-25 cm of the soil profile, where the majority of soil-nutrient-plant interactions are expected to occur (Sharpley *et al.* 1996, Rayment and Lyons 2011). However, this depth may vary according to the nature of the crop and or type of assessment (Rayment and Lyons 2011). For example: for the assessment of P fertiliser requirements for dairy pastures it is common to sample soils at 0-10 cm (Rayment and Lyons 2011).

Major variations in soil type and pH make it difficult for any single soil P test to be accurate for all soils and hence there are a range of agronomic P test methods available (Mason *et al.* 2013). Some of the more common testing methods may include: Colwell-P, Mehlich-3 P, Bray-1 P and Olsen-P (Wang *et al.* 2015). Tests such as these are often collectively referred to as soil test phosphorus (STP) (Sims and Sharpley 2005, Wang *et al.* 2015).

One of the major differences among common agronomic soil P tests is the chemical composition of reagents used in the P extraction process (i.e., extractants) (Mason *et al.* 2013). Methods which incorporate acidic regents (e.g. Bray-1 P and Mehlich-3 P) will extract different amounts of P from the same soils compared to methods using alkaline reagents (e.g. Olsens-P and Colwell-P) (Mason *et al.* 2013). Therefore, some methods are preferred over others in certain regions based on their suitability to the general character of soils within that area (Bolland and Gilkes 2004, Abdala *et al.* 2015). For example, the preferred method in Australia is Colwell-P, whereas in the USA, the favoured methods are Mehlich-3 P (in the Mid-Atlantic regions), Bray-1 P and Olsen P (Bolland and Gilkes 2004, Abdala *et al.* 2015).

More recently, phosphorus buffering index (PBI) has also often been included in the suite of routine agronomic soil tests as it has been shown that it can be used to assess the bioavailability of nutrient inputs from fertiliser (Moody 2007).

Environmental tests

Sorption capacity

The sorption potential (i.e., sorption strength and capacity) of soils can be derived from a sorption isotherm (Bache and Williams 1971, Holford *et al.* 1997, Sims and Sharpley 2005). Sorption isotherms can be generated experimentally by mixing (e.g. via shaking) a soil with a range of standardised P solutions with increasing concentrations (Holford *et al.* 1997). A sufficient shaking period is generally determined by estimating the time required to reach an 'equilibrium' concentration (i.e., the point where no net sorption or desorption is occurring) (Holford *et al.* 1997). After shaking, the solution P concentration is measured and the amount of P removed from the original solution is used to construct the sorption isotherm (Holford *et al.* 1997). Thus, the decrease in soil solution P is plotted against the original concentrations, which commonly results in an initial steep gradient followed by a gradual plateau, where the soils sorption capacity nears exhaustion (Figure 1) (Bache and Williams 1971). Hence, P sorption isotherms tend to be somewhat asymptotic (Holford *et al.* 1997). Therefore, realistic maximum values representing the point at which the soil has reached its "sorption capacity" are not normally observed and thus a value is often calculated mathematically by fitting a suitable equation (Bache and Williams 1971, Holford *et al.* 1997). The two most common mathematical models used for this purpose are the linear, Langmuir and Freundlich equations (Sims and Sharpley 2005, Vaananen *et al.* 2008, Wu and Sansalone 2013). The suitability of each of these models may vary among different soil types (Singh and Gilkes 1991, Ahmed *et al.* 2008, Perez *et al.* 2014).

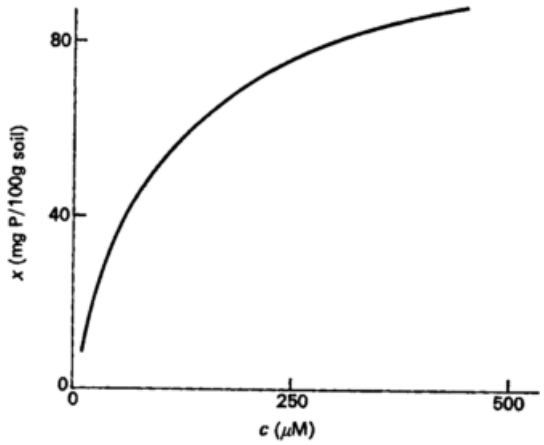


Figure 1. Example of a typical P sorption curve where P sorbed (x) is plotted against the equilibrium concentration (c) results in a sharp incline in sorption followed by a plateau as the soil approaches sorption capacity (figure from Bache and Williams 1971).

More recently, advancements made through method development strategies has given rise a number of other various tests used to quantify sorption capacity of a soil, many of which have been designed to be included into the general suite of agronomic testing, One example is the phosphorus buffering index (PBI) test, which has since been found to be strongly correlated with nutrient availability. The PBI test is essentially a one point sorption curve (Weaver and Wong 2011).

Water extractable phosphorus

Water extractable P (WEP) is a test designed to predict the amount of P lost from soils in leachate or runoff (Sims and Sharpley 2005, Wang *et al.* 2015). Therefore, WEP is most commonly determined via some form of extraction experiment, involving a given amount of soil mixed with a solution of a certain ionic strength (Sims and Sharpley 2005). The mobilisation of P from solid to aqueous phase is influenced by a combination of the ionic strength and ratio of soil solution to the soil (Sims and Sharpley 2005). Hence, extraction methods may vary based on the environmental conditions or scenario attempted at being imitated (Sims and Sharpley 2005). For example: high (10,0000:1) soil solution to soil ratios may be used to simulate runoff interactions whereas low ratios (1:1) may be used to simulate leaching conditions (Sims and Sharpley 2005). Additionally, the chemical composition of the extractant is often altered to simulate various environmental conditions; for example: using a dilute salt solution (e.g., 0.01M $CaCl_2$) when predicting potential P lost from intense storms of limited duration (Sharpley *et al.* 1996, Sims and Sharpley 2005).

DGT

Recent research efforts have demonstrated that there is potential for DGT (diffusive gradients in thin-films), for determining plant available P in soil (Dougherty *et al.* 2011, Mason *et al.* 2013). Unlike other soil testing methods, the DGT method may be suitable for all soil types, and may therefore be universally applicable for agricultural soil P assessments (Mason *et al.* 2013).

This method is carried out by placing a DGT device in contact with a soil for a given period of time (Dougherty *et al.* 2011). During this time, P accumulates on the DGT device (binding layer) as a function of both the concentration of P in the soil solution (i.e., soil pore-water) and the rate which P is being desorbed into the soil solution (Dougherty *et al.* 2011). The DGT method is therefore designed to imitate the uptake of soluble P by plant

roots by providing a sink for the free orthophosphate ions (Mason *et al.* 2013). A major advantage of this method is that it provides useful means for obtaining data under conditions that are close to those experienced by the plant root during P uptake (i.e., field conditions) (Mason *et al.* 2013).

Iron-oxide-coated filter paper strip-extractable phosphorus (FeO-P)

The iron-oxide coated filter paper strip P (FeO-P) test was originally designed for agronomic purposes, as it provides a measure of dissolved reactive P (DRP) which has been closely linked to concentrations of bioavailable P in soils (Pote *et al.* 1996, Wang *et al.* 2010). However, FeO-P has also been widely used as a test for predicting the potential concentration of DRP in runoff following a rainfall event (Wang *et al.* 2015). FeO-P is, therefore, often used for carrying out soil P loss risk evaluation experiments (Wang *et al.* 2015).

Degree of phosphorus (sorption) saturation

The degree of phosphorus sorption saturation DPSS is calculated as the ratio (as a percent) between sorbed P and the P sorption capacity, as shown in the equation below (Wang *et al.* 2015):

$$\text{DPSS}\ (\%) = \frac{\text{Sorbed P}}{\text{P sorption capacity}} \text{x}\ 100$$

DPSS can provide a good indication of P mobility in soils and is often used as an environmental index for assessing the potential for P to be lost via runoff and leaching (Ige *et al.* 2005, Sims and Sharpley 2005, Simmonds *et al.* 2015). DPSS is often referred to in the literature in a slightly abbreviated form as DPS (degree of phosphorus saturation), but it represents the same parameter (Ige *et al.* 2005, Sims and Sharpley 2005, Wang *et al.* 2015).

There are many variations as to how this parameter may be calculated, as each of the components used in the equation can be determined by a range of methods (Ige *et al.* 2005). Recently it has been shown that DPS can be derived from common agronomic STPs or FeO-P and an associated sorption index (Ige *et al.* 2005). In addition, various tests which have proven to be useful for indicating sorption capacity have also been used to calculate DPS, negating the need to generate a sorption isotherm (Wang *et al.* 2010, Wang *et al.* 2015). Such measurements include, water extractable P (WEP), aluminium (Al) and iron

(Fe) oxyhydroxide extractable P (for acid soils) or (Ca + Mg) Mehlich 3 in calcareous soils (Ige *et al.* 2005). Hence, it is important that the methods used to derive DPS are chosen carefully according to the soil type, for example, FeO-P may be used as in indicator of sorbed P in acidic soils but for alkaline calcareous soils it would not be as representative (Ige *et al.* 2005, Simmonds *et al.* 2015).

DPS has been reported as a suitable parameter to set environmental limits for soil P, as it has been found to be closely correlated with water extractable P (WEP) (Maguire and Sims 2002, Ige *et al.* 2005, Sims and Sharpley 2005, Wang *et al.* 2015). Research has shown that this correlation generally follows a two-step pattern where after a certain DPS level (i.e., change point), any further additions of P will become highly likely to be lost to receiving water bodies via runoff or leaching (Maguire and Sims 2002). For example, in the Netherlands, this limit is set at 25%, therefore, farms are legally required to maintain soil DPS < 25% (Ige *et al.* 2005, Sims and Sharpley 2005). It has been shown, however, that appropriate thresholds may vary for different soil types depending on where the change point occurs (Khiari *et al.* 2000).

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